# *Factors Influencing Success Of Cutthroat Trout Translocations*

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*EXECUTIVE SUMMARY*

# FACTORS INFLUENCING SUCCESS OF CUTTHROAT TROUT TRANSLOCATIONS

Native subspecies of cutthroat trout (*Oncorhynchus clarki*) in the western United States have experienced drastic declines in their distributions, often to <5% of their native range, due to habitat degradation, the introduction of nonnative salmonids, and overharvest. Of the 14 subspecies recognized (three are undescribed), one is extinct, four are listed as threatened or endangered under the Endangered Species Act, and conservation plans have been developed for most others. Cutthroat trout readily hybridize with other spring-spawning salmonids and are apparently displaced by fall-spawning species such as brook char (*Salvelinus fontinalis*) and brown trout (*Salmo trutta*), so many populations are restricted to remote headwater streams and lakes above fish movement barriers that prevent invasion by nonnative salmonids. Isolating barriers, while protecting cutthroat trout populations from nonnative salmonids, restrict them to areas that may be too small or have insufficient habitat to support a viable population.

Establishing new cutthroat trout populations through translocation of genetically pure fish into fishless waters or those treated with toxicants to remove nonnative salmonids is one of the few management strategies for increasing their range. Unfortunately, success rates for establishing self-sustaining fish populations through translocation are generally less than 50%, and few recovery programs have reviewed their translocation attempts to determine the factors influencing translocation success in aquatic systems. Conservation of greenback cutthroat trout (*O. c. stomias*), one of the

listed subspecies, and Rio Grande cutthroat trout (*O. c. virginalis*), a subspecies that has been petitioned for federal listing, has been ongoing for more than 25 years. More than 40 translocations have been conducted for each subspecies, so they represent two of the few cases where recovery efforts for any fish are sufficient to evaluate factors influencing success. ***Our research objective was to identify factors that promote establishment and persistence of cutthroat trout populations isolated in headwater streams by comparing attributes of translocations that successfully established a naturally reproducing cutthroat trout population to those that failed.***

When we began this research in 1995, biologists that manage native cutthroat trout in the Central Rocky Mountains had many different hypotheses for why certain translocations failed, including inadequate habitat area, low habitat quality, high stream gradient, cold water temperatures, and excessive angling mortality. Furthermore, data on populations of Rio Grande and greenback cutthroat trout were dispersed among miscellaneous agency reports, conservation and recovery plans, computerized stocking records, and the personal files of managers and biologists. We began our research by summarizing the status of individual populations based on a compilation of these agency data. For Rio Grande cutthroat trout, we prepared a project report in 1996, titled *Compilation of Data on Colorado Waters Containing Rio Grande Cutthroat Trout* (Harig and Fausch 1996a), which presented the data and provided a simple summary of population status. For greenback cutthroat trout, we conducted a series of more extensive analyses of the agency data (e.g., Harig and Fausch 1996b) and present the results in Chapter I of this final project report. We analyzed unpublished data from natural resource agencies on reinvasion of nonnative salmonids, potential limiting habitat factors,

and source of translocated fish to identify the attributes of the 14 of 37 translocations of greenback cutthroat trout that were successful, and the probable cause of failure for the other 23. Of the 23 that failed, 11 (48%) were reinvaded by nonnative salmonids, 10 (43%) apparently had unsuitable habitat, and 2 were depressed by other factors. Reinvasion occurred most often because of incomplete removal of nonnative salmonids in complex habitats or failed artificial barriers. ***Of those greenback cutthroat trout translocations not reinvaded by brook trout, success was highest in receiving waters with at least 2 ha of habitat that previously supported reproducing trout populations.***

In Chapter II, we report on a detailed analysis of the minimum habitat required for establishment and persistence of translocated native cutthroat trout populations. The analysis is based on four years of extensive field surveys of stream-scale habitat, and map measurements of basin-scale habitat, for 27 greenback and Rio Grande cutthroat trout translocations in Colorado and New Mexico (Harig and Fausch 1997, 1998, 1999). The best models developed from these data using polytomous logistic regression predict risk of translocation success or failure from stream-scale habitat attributes. ***These models indicate that cold summer water temperature, narrow stream width, and lack of deep pools limit translocated populations of native cutthroat trout.*** Cold summer temperatures are known to delay spawning and prolong egg incubation, which reduces the growth of fry and likely limits their overwinter survival. Furthermore, small streams with few deep pools may lack the space necessary to promote overwinter survival of a sufficient number of individuals to sustain a viable population.

***Managers and biologists can use this stream-scale habitat model to evaluate potential translocation sites and identify current populations at greatest risk from***

***extirpation (Figures 1-3).*** The probability that a translocation stream will support a high abundance of cutthroat trout is calculated as:

$$P(high) = 1 - \left[\frac{\exp(14.077 - 0.891t - 1.451w - 0.017d)}{1 + \exp(14.077 - 0.891t - 1.451w - 0.017d)}\right]$$

where $P$ = probability of a stream having a population status of high, $t$ = mean daily water temperature for July ($^{o}$C), $w$ = mean bankfull width of pools (m), and $d$ = total number of deep pools (residual depths ≥ 30 cm; Figures 1). For example, stream habitat was measured in Powderhouse Creek, a site where the translocation was too recent (1997) to assess success, to demonstrate use of this model. Powderhouse Creek has a mean daily July water temperature of 10.0 °C, mean pool bankfull width of 2.2 m, and 5 deep pools. Given this, the model predicts only a 13% chance of establishing a translocated population with high numbers of cutthroat trout:

$$P(high) = 1 - \left[\frac{\exp(14.077 - 0.891(10.0) - 1.451(2.2) - 0.017(5))}{1 + \exp(14.077 - 0.891(10.0) - 1.451(2.2) - 0.017(5))}\right]$$

$$P(high) = 1 - \left[\frac{\exp(1.890)}{1 + \exp(1.890)}\right] = 1 - \left[\frac{6.618}{7.618}\right] = 1 - 0.869 = 0.131$$

A translocation into Powderhouse Creek is more likely to establish low numbers of trout (54% probability) or no trout (33% probability; see Chapter II). Therefore, managers will need to decide whether establishing a small population of cutthroat trout is worth the time, money, and effort of a translocation project, particularly considering that small populations are at greater risk of extirpation from demographic and environmental stochasticity.

These models can be used to make predictions about the potential for success of future cutthroat trout translocations into similar stream habitats (Figures 2 and 3). For

example, in streams with the average number of pools for the streams we measured (87 pools), the chances of establishing a population with high numbers of cutthroat trout by translocation into a narrow stream (2.0 m wide) is >50% only if mean July water temperatures exceed about 11 °C, whereas in wider streams (4.0 m wide) chances are >50% if water temperatures exceed about 8 °C (Figure 2). Similarly, in streams of the average width for those we measured (3.4 m), chances of establishing a high population by translocation are >50% for cold streams (7 °C mean July water temperature) only when about 200 or more deep pools are available, whereas chances of success are >50% for warmer streams (10 °C mean July temperature) when 50 or more deep pools are present. We caution that these models are not accurate when extrapolating beyond the range of data used to develop them (see Chapter II), and should be used only as guidelines because they are based on observational data from streams not randomly selected by managers rather than a controlled and replicated experimental manipulation.

***Models of basin-scale habitat were not as effective as stream-scale habitat for distinguishing between successful and unsuccessful translocations of cutthroat trout, but do predict that streams with watersheds >14.7 km² have at least a 50% chance of supporting a high population of cutthroat trout.*** This minimum watershed area criterion may be useful as a coarse filter for selecting potential translocation streams or for identifying historical populations at the greatest risk of extirpation. We caution that the habitat attributes necessary to establish a population through translocation may not be identical to those that have sustained historical populations. However, managers have limited time and budgets to devote to stream surveys so a coarse filter such as minimum watershed area could be used to prioritize streams. Presumably, large watersheds

($> 14.7$ km$^2$) encompass low elevation habitat that provides warmer summer temperatures, and would have relatively wide stream channels of sufficient length to provide an adequate number of deep pools. At last count, there were 50 streams with historical, remnant populations of Rio Grande cutthroat trout in New Mexico that remained free from nonnative salmonids. Almost half (24) of these 50 populations are in watersheds $\leq 14.7$ km$^2$ in area and may be at risk of extirpation from insufficient habitat. Similarly, in Colorado 4 of 13 historical populations of Rio Grande cutthroat trout and 4 of 7 historical populations of greenback cutthroat trout also persist in small watersheds $<$ 14.7 km$^2$. Field surveys of stream-scale habitat could identify if these small watersheds have appropriate summer water temperatures, and are wide enough and have a sufficient number of deep pools to support a high abundance of cutthroat trout.

This research is one of the few attempts to determine the specific factors influencing translocation success for fishes, and it demonstrates that measuring attributes of local habitat over a whole watershed scale that matches the life history of the organism can be highly useful for identifying critical habitat factors. ***The models we developed from stream- and basin-scale habitat attributes will be valuable tools for fisheries managers concerned with the conservation of greenback and Rio Grande cutthroat trout, particularly if included in an active management program that tests and refines these models with data from recent and future translocation sites.*** Moreover, a coarse-scale analysis of this type may also be applicable to other subspecies of cutthroat trout in central and southern Rocky Mountain streams (e.g., Colorado River cutthroat trout, *O. c. pleuriticus*) because similar habitat attributes probably limit their populations.

REFERENCES


Harig, A. L., and K. D. Fausch. 1996a. Compilation of data on Colorado waters containing Rio Grande cutthroat trout. Annual Report to Colorado Division of Wildlife, Montrose, CO.

Harig, A. L., and K. D. Fausch. 1996b. Developing incidence functions for cutthroat trout: determining the risk of extinction for isolated populations. Annual Report to U.S. Forest Service Rocky Mountain Fish Habitat Relations Program, Laramie, WY.

Harig, A. L., and K. D. Fausch. 1997. Evaluating risks of extinction from habitat fragmentation for native cutthroat trout in Colorado and Wyoming. Annual Report to Colorado Division of Wildlife, U.S. Forest Service, and Trout Unlimited.

Harig, A. L., and K. D. Fausch. 1998. Evaluating risks of extinction from habitat fragmentation for native cutthroat trout in Colorado. Annual Report to Colorado Division of Wildlife, U.S. Forest Service, and Trout Unlimited.

Harig, A. L., and K. D. Fausch. 1999. Minimum habitat requirements for establishing translocated cutthroat trout populations. Annual Report to Colorado Division of Wildlife, U.S. Forest Service, and Trout Unlimited.

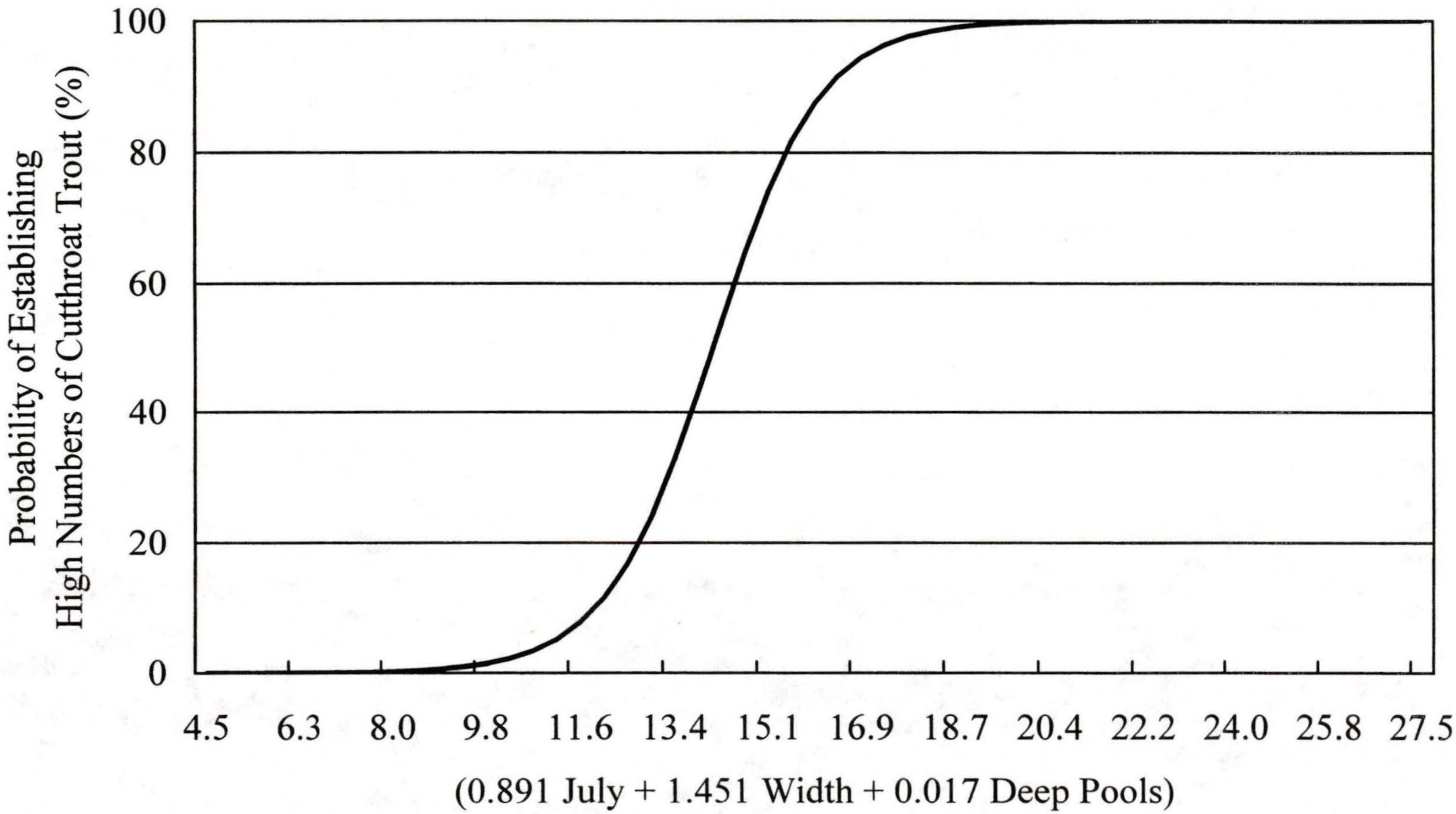


Figure 1. The "best" model for predicting the probability of establishing high numbers of cutthroat trout in a potential translocation stream based on the mean daily water temperature for July (degrees C), mean bankfull pool width (m), and number of deep pools (residual depth > 30 cm).

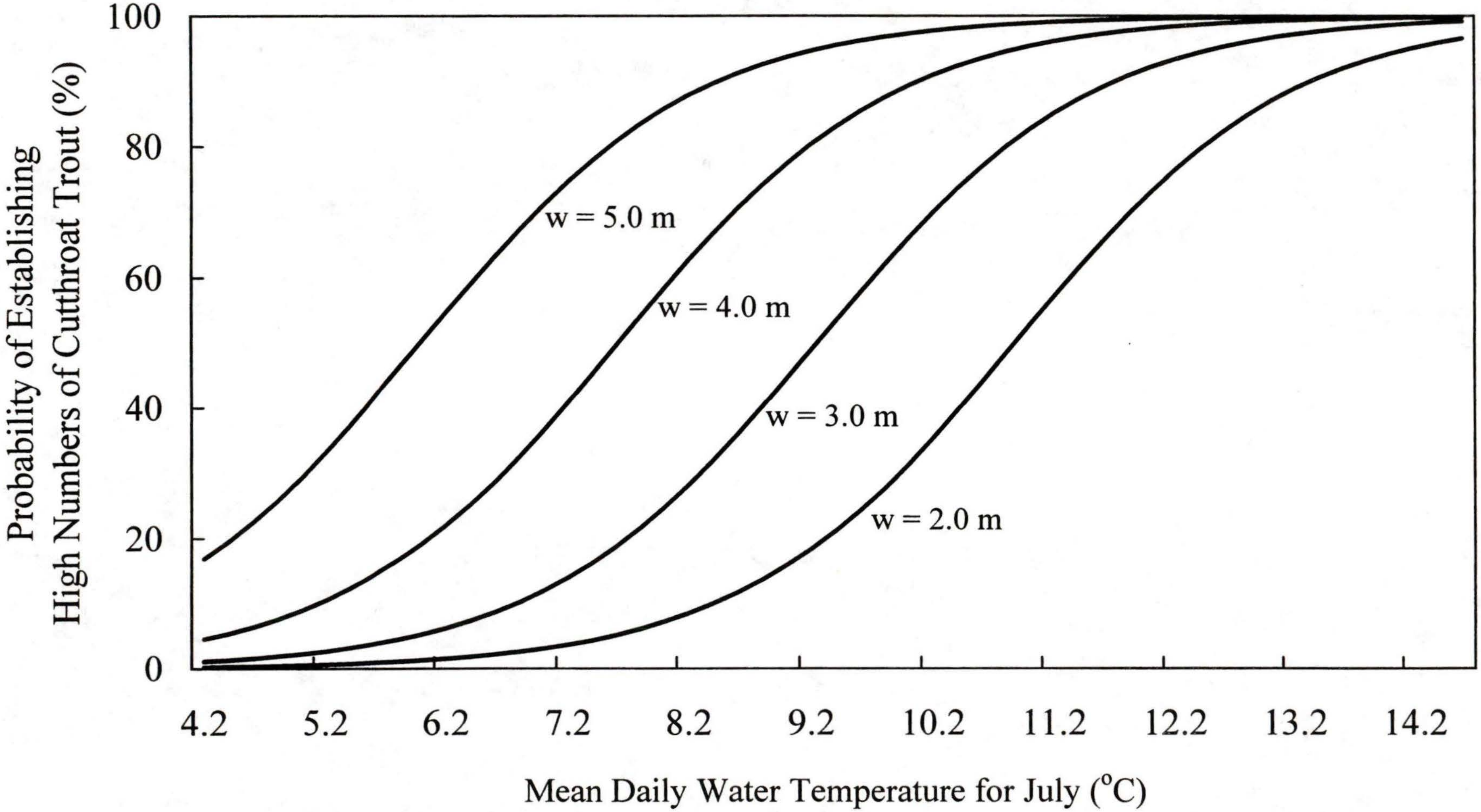


Figure 2. Predicting the probability of establishing high numbers of cutthroat trout in a potential translocation stream based on the mean daily water temperature for July (degrees C) for streams with an average number of deep pools (87 pools with a residual depth > 30 cm) and mean bankfull pool width (w) of 2.0, 3.0, 4.0, or 5.0 m.

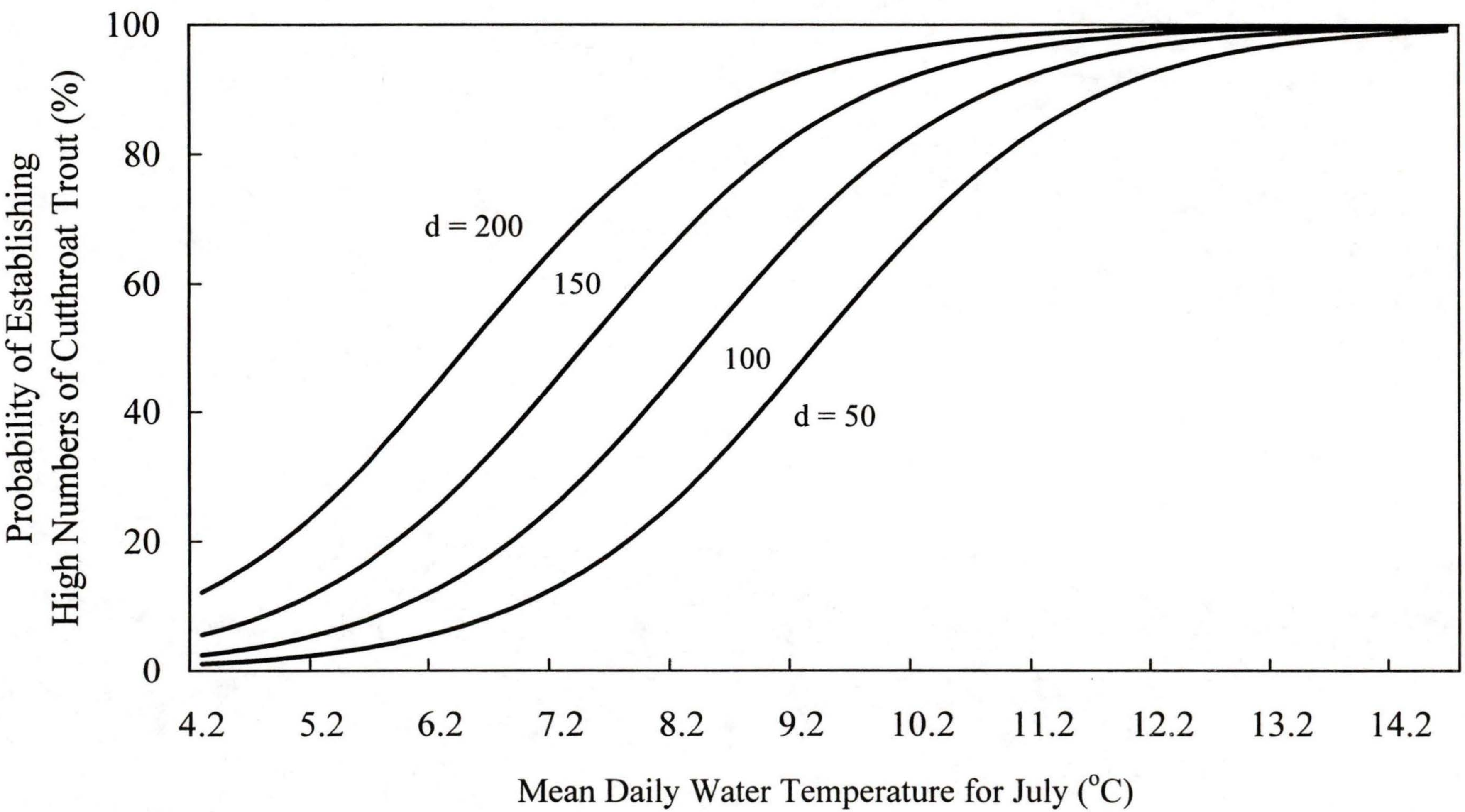


Figure 3. Predicting the probability of establishing high numbers of cutthroat trout in a potential translocation stream based on the mean daily water temperature for July (degrees C) for streams with an average bankfull pool width (3.4 m) and 50, 100, 150, or 200 deep pools (d; residual depth > 30 cm).

TABLE OF CONTENTS

Page

CHAPTER I: Factors Influencing Success Of Translocated Greenback Cutthroat Trout Populations........ 1

ABSTRACT........ 1

INTRODUCTION........ 2

HISTORY OF GREENBACK CUTTHROAT TROUT RECOVERY........ 3

METHODS........ 4

RESULTS AND DISCUSSION........ 6

Nonnative Salmonids........ 7

Habitat Size and Quality........ 8

Source of Translocated Fish........ 10

Conclusions........ 12

ACKNOWLEDGMENTS........ 12

REFERENCES........ 13

TABLE 1.1........ 19

FIGURE 1.1........ 20

CHAPTER II: Minimum Habitat Requirements For Establishing Translocated Cutthroat Trout Populations........ 22

ABSTRACT........ 22

INTRODUCTION........ 23

SALMONID HABITAT AND POTENTIAL LIMITING FACTORS........ 28

Stream-Scale Habitat........ 28

Basin-Scale Habitat........ 30

Page

METHODS........................................................................ 32

Study Sites........................................................................ 32

Field Surveys of Stream-Scale Habitat and Cutthroat Trout Abundance.. 34

Measurement of Basin-Scale Habitat using Digital Data.................. 36

Statistical Models of Translocation Success..................................37

RESULTS........................................................................ 40

Relative Population Status........................................................ 40

Models of Translocation Success Based on Water Temperature......... 41

Models of Translocation Success Based on Water Temperature and Stream-Scale Habitat........................................................ 42

Models of Translocation Success Based on Basin-Scale Habitat..........44

Models of Translocation Success Based on Stream- and Basin-Scale Habitat........................................................................ 46

DISCUSSION........................................................................ 46

Stream Habitat Attributes Limiting Cutthroat Trout....................... 46

Appropriate Spatial Scales for Isolated Trout Populations................ 48

Probability of Long-term Persistence........................................ 52

Management Implications.........................................................53

ACKNOWLEDGMENTS........................................................... 55

REFERENCES........................................................................56

TABLES........................................................................ 67

FIGURES........................................................................ 84

Page

APPENDIX I: Greenback Cutthroat Trout Translocation Projects........................ 92

APPENDIX II: Agency Stocking Records for Greenback and Rio Grande Cutthroat Trout Translocation Streams............................................... 97

APPENDIX III: Values for Remaining Stream-Scale Habitat Variables................ 101

APPENDIX IV: Values for Remaining Basin-Scale Habitat Variables................. 105

APPENDIX V: Values for Basin-Scale Habitat Variables for Historical Stream Populations of Greenback and Rio Grande Cutthroat Trout...................... 109

APPENDIX VI: Thermograph Data from 1998-1999..................................... 115

# CHAPTER I:

# FACTORS INFLUENCING SUCCESS OF TRANSLOCATED GREENBACK CUTTHROAT TROUT POPULATIONS


## ABSTRACT

Native subspecies of cutthroat trout (*Oncorhynchus clarki*) have declined drastically due to the introduction of nonnative salmonids, overharvest, and habitat degradation. Conservation of most declining subspecies will include establishing new populations through translocation of genetically pure fish. Recovery of greenback cutthroat trout (*O. c. stomias*) has been ongoing for 25 years, so their translocation attempts provide unique empirical information to guide recovery of other non-anadromous salmonids. We compared 14 translocation projects that established populations of greenback cutthroat trout to 23 that failed to determine factors that influence translocation success. Of the latter, 11 (48%) were reinvaded by nonnative salmonids, 10 (43%) apparently had unsuitable habitat, and 2 were depressed by other factors (heavy metal pollution or bird predation). Reinvasion occurred most often because of incomplete removal of nonnative salmonids in complex habitats or failed artificial barriers. Of those not reinvaded, success was highest in receiving waters with at least 2 ha of habitat that previously supported reproducing trout populations.

## INTRODUCTION

Native subspecies of cutthroat trout (*Oncorhynchus clarki*) in the western United States have experienced drastic declines in their distributions, often to <5% of their native range, due to habitat degradation, the introduction of nonnative salmonids, and overharvest (Gresswell 1988; Behnke 1992; Young 1995). Of the 14 subspecies recognized (Behnke 1992; three are undescribed), one is extinct, four are listed as threatened or endangered under the Endangered Species Act (ESA), and conservation plans have been developed for most others. Several of the latter have been petitioned for federal listing.

Primary management strategies in the conservation of declining cutthroat trout subspecies include identifying and maintaining remnant populations and establishing new populations through translocation of genetically pure fish. However, managers' ability to implement effective translocation projects is currently limited by lack of experimental or observational data on what influences successful establishment of a population. Recovery of greenback cutthroat trout (*O. c. stomias*), one of the listed subspecies, has been ongoing for more than 25 years, and managers have attempted 44 translocations. This represents one of the few cases where recovery efforts for any fish are sufficient to evaluate factors influencing success (Williams et al. 1988; but see Simons et al. 1989). Our objective is to analyze such factors for the greenback cutthroat trout translocations, and thereby seek lessons to improve conservation of other non-anadromous salmonids.

## HISTORY OF GREENBACK CUTTHROAT TROUT RECOVERY

Greenback cutthroat trout were once abundant in the South Platte and Arkansas river basins of Colorado and Wyoming (Figure 1.1; Behnke 1992), but were decimated after 1860 by overfishing and habitat degradation from mining, railroad construction, and agriculture (Jordan 1891; Wiltzius 1985). Unfortunately, the few early attempts to culture greenback cutthroat trout failed (Dwyer and Rosenlund 1988), so nonnative brook trout (*Salvelinus fontinalis*) and rainbow trout (*O. mykiss*) were introduced widely after railroad lines reached Denver in 1870 (Wiltzius 1985). Greenback cutthroat trout readily hybridize with spring-spawning rainbow trout, are apparently displaced by fall-spawning brook trout and brown trout (*Salmo trutta*; Wang and White 1994), and are more vulnerable to angling than these species (MacPhee 1966; Behnke 1992). Consequently, the remaining greenback cutthroat trout populations are restricted to small remote, high-elevation streams and lakes above fish movement barriers. Many of these habitats are cold and unproductive and undergo extreme flow fluctuations, leading to small, slow-growing trout populations (cf. Young 1995). Thus, the two main biological constraints hampering greenback cutthroat trout populations are invasions by nonnative salmonids and lack of high quality habitat.

Though believed extinct by the 1930s (Green 1937), greenback cutthroat trout were discovered in two streams in 1965 and 1970 and listed as endangered under the ESA in 1973 (USFWS 1998). Listing initiated a formal multi-agency recovery effort to restore the greenback cutthroat trout to a portion of its native range by 2000 (Stuber et al. 1988; USFWS 1998). A primary management objective in recovery is to establish new populations by translocating genetically pure greenback cutthroat trout into fishless

waters or those treated with toxicants to remove nonnative salmonids. Of 44 translocations since 1965, 37 were before 1991, which is long enough to assess whether they were successful in establishing a self-sustaining cutthroat trout population (USFWS 1998). No new translocation projects were conducted during 1991-1994 due to lack of funding, and seven translocations conducted after 1995 are too recent to assess population status.

## METHODS

We identify factors influencing success of the 37 greenback cutthroat trout translocations conducted prior to 1995 by comparing translocations that produced self-sustaining trout populations to those that failed. Translocations were considered successful if they established a population that met the four quantitative criteria defined in the greenback cutthroat trout recovery plan (USFWS 1998):

1. Maintain a minimum biomass of 22 kg/ha through natural reproduction;
2. Support at least 500 adult greenback cutthroat trout (>120 mm total length);
3. Produce at least two year-classes by natural reproduction within 5 years; and
4. Be isolated from nonnative salmonids.

The recovery plan also states that more than one population can occupy a drainage fragmented by natural barriers to upstream fish movement if each habitat fragment contains at least 2 ha of habitat and meets the four recovery criteria (USFWS 1998). Although a minimum habitat requirement was applied only to populations in fragmented systems, it could also be used to assess habitat for other greenback cutthroat trout populations (Dwyer et al. 1994). Unfortunately, few data are included in the recovery

plan or other reports to independently evaluate whether the translocated populations meet any of the criteria. Although some data on standing stock and habitat size are included in the recovery plan, none are available for number of adults, success of year classes, or isolation from nonnative salmonids. Therefore, we assume that translocations considered successful in the recovery plan meet all criteria. Translocations were considered unsuccessful if they failed to meet one or more of the criteria.

For each translocation, we identified population status (successful or unsuccessful), type of translocation (into previously barren waters or those treated with chemical toxicants to remove nonnative salmonids), source of translocated fish (hatchery or wild), location and type of fish movement barrier, median elevation, habitat area, and probable cause of failure, if applicable (see Appendix I). Data were collected from the greenback cutthroat trout recovery plan (USFWS 1998), agency reports (Mullan 1973; Behnke and Zarn 1976; Hickman and Miller 1977; Langlois et al. 1978, 1979; Washington 1981; Culver and Lentsch 1982; Culver and Bestgen 1983, 1985; Culver 1983; Davis and Culver 1984; Chart et al. 1986, 1987; Stevens and Rosenlund 1986; Kehmeier et al. 1988; Rosenlund and Stevens 1988, 1990; Kehmeier and VanBuren 1990; Rosenlund et al. 1994; Winters et al. 1995), and personal communications with the Greenback Cutthroat Trout Recovery Team. Some explanatory variables were further evaluated with a two-sided Fisher's exact test (Agresti 1996; S-PLUS 4 1997). This is a conservative statistical test, so P-values were assessed as a measure of their strength of evidence rather than choosing an arbitrary significance level (Agresti 1996). However, translocation sites were not chosen randomly by managers, so statistical results imply

only an association between success and significant factors, not a direct cause-and-effect relationship.

## RESULTS AND DISCUSSION

Fourteen of the 37 translocations of greenback cutthroat trout were considered successful in establishing populations according to recovery plan criteria (USFWS 1998). This translocation success rate (38%) is similar to that reported for threatened and endangered birds and mammals worldwide (44% [n=80]; Griffith et al. 1989), and much higher than for one fish (18% [n=191] for Gila topminnow, *Poeciliopsis occidentalis occidentalis*; Simons et al. 1989). However, translocation success was somewhat lower than reported for other non-anadromous salmonids. Of those for which sufficient time was available to assess population status, 80% of Gila trout (*O. gilae*) translocations (n=5; Propst et al. 1992), and 83% of Bonneville cutthroat trout (*O. c. utah)* translocations (n=6; Hepworth et al. 1997) were considered self-sustaining or expanding. Nonetheless, sample sizes were small for both translocation programs, and criteria for defining success may have been less stringent than the greenback cutthroat trout recovery criteria. Of the 23 unsuccessful translocations of greenback cutthroat trout, 11 (48%) were reinvaded by nonnative salmonids, 10 (43%) apparently had unsuitable habitat, and 2 were depressed by other factors (heavy metal pollution or bird predation; USFWS 1998). We address the first two factors in greater detail.

*Nonnative Salmonids*

Nonnative salmonids are probably the greatest threat to the establishment of cutthroat trout populations because nearly all are vulnerable to natural invasion or human transplant from downstream. Empirical data show that nearly half of translocations into waters with supposedly suitable habitat failed due to reinvasion by nonnative salmonids (11 of 25; other 12 waters excluded due to unsuitable habitat or other factors). Five translocations were unsuccessful apparently because of incomplete removal of nonnative salmonids during a single treatment with fish toxicants (rotenone or antimycin). Complex habitats such as beaver ponds, springs, bogs, and multiple tributaries were thought to have provided refugia that allowed some fish to survive, because nonnative salmonids were later found concentrated in these areas. The probability of achieving a complete fish kill during chemical reclamation might be improved by treating with toxicants for multiple years and monitoring for nonnative salmonids before introducing the native trout (cf. Rinne et al. 1981; Propst et al. 1992). Nonetheless, one-time chemical treatments were successful in eliminating nonnative salmonids from 13 waters.

Four translocations were thought to have failed when brook trout, brown trout, or rainbow trout breached artificial barriers to fish movement, because the invading species were found just upstream from the barriers. Nonnative salmonids breached rock gabions and wooden dams either by surmounting structures (heights 1-3 m) or through channels eroded around or beneath the barrier. No artificial barrier except an 18-m high reservoir spillway successfully prevented upstream movement of nonnative salmonids, so choosing only natural barriers unlikely to fail could reduce the risk of translocation failure. All

other successful greenback cutthroat trout populations were above natural waterfalls, steep cascades, or stream reaches with temperatures too warm for trout survival.

Managers suspect that at least two translocations failed because anglers deliberately reintroduced brook trout. Brook trout in these systems were concentrated around popular campsites and trail crossings, and anglers had threatened to restock brook trout after chemical treatments (B. Rosenlund, U.S. Fish and Wildlife Service, personal communication). Deliberate introduction of nonnative salmonids by anglers does not have a simple solution. Selecting either remote locations or those that can be closed to motorized traffic reduces potential human intervention, but in the past has focused recovery efforts on small, high-elevation, unproductive habitats. Establishing populations on private property may provide low-elevation, productive habitats in a protected environment (T. Nesler, Colorado Division of Wildlife, personal communication).

*Habitat Size and Quality*

The 10 unsuccessful translocations not invaded by nonnative salmonids or depressed by other factors were assumed to have failed due to unsuitable habitat. The small size and low productivity of the headwater sites chosen for most translocations suggests that either habitat size or quality may be limiting cutthroat trout populations. Comparison of these 10 translocations to successful ones showed that success in establishing translocated populations was higher in receiving waters with sufficient minimum habitat area (ca. 2 ha) that previously supported reproducing trout populations (Table 1.1), similar to results for birds and mammals worldwide (Griffith et al. 1989).

Our analysis combined with information from managers suggests that lakes $\geq 2$ ha with inlets or outlets that warm early and have sufficient baseflow for fry rearing have the highest probability of supporting populations. These larger habitats may buffer individual greenback cutthroat trout populations from environmental fluctuations and reduce their risk of extirpation from floods, wildfires, and droughts (Propst et al. 1992). Streams may also be suitable if long enough to encompass sufficient habitat. For example, mean bankfull stream widths from recent surveys of greenback cutthroat trout translocations indicate that streams at least 5.7-km long are needed to provide 2 ha of habitat at bankfull flows. Only one translocation that failed due to unsuitable habitat was a stream longer than 5.7 km, thus many of the shorter streams might have had insufficient space to sustain adults and juveniles. Native trout translocated to smaller habitats such as these will likely depend on continued management to maintain population persistence (Lubow 1996; Foin et al. 1997).

Translocation attempts in previously barren waters were less successful than projects where nonnative trout were successfully removed using toxicants (Table 1.1), even though many of the latter were subsequently reinvaded. Barren waters may provide high quality habitat if downstream barriers prevented colonization, but it is likely that many lacked trout due to unsuitable habitat. For example, some lakes lacked inlets or outlets with clean gravel that could support spawning, although translocation into one of these lakes was primarily for angling opportunities to engender public support for restoration (USFWS 1998). Translocations to barren waters may also endanger aquatic fauna not adapted to trout predation or competition, such as amphibians or large macroinvertebrates (Harig and Bain 1998, Tyler et al. 1998).

Elevation apparently had little direct effect on translocation success (Table 1.1), although experimental stocking of greenback cutthroat trout in one lake at 3511 m showed they failed to reproduce (B. Rosenlund, U.S. Fish and Wildlife Service, personal communication). Nearly all waters where greenback cutthroat were translocated before 1995 are at high elevation (70% >2800 m, 40% >3000 m; n=37), and within the elevation range of 1900-3500 m where translocations were attempted, there is no apparent relationship between elevation and population status. However, it is likely that low summer water temperatures, which are strongly influenced by factors other than elevation (Smith and Lavis 1975), may have inhibited reproductive success (Harig and Fausch unpublished). In streams where water temperatures do not reach 4-8°C by early July, greenback cutthroat trout spawning is delayed (Behnke 1992; USFWS 1998) and egg incubation is prolonged, leading to low embryo survival and late hatching (Rinne 1980; Hubert et al. 1994; Stonecypher et al. 1994; Thurow and King 1994). The smaller fry that result risk overwinter starvation if they cannot grow enough to withstand metabolic deficits at low winter temperatures (Hunt 1969; Cunjak and Power 1987; Shuter and Post 1990). Thus, the greater success of translocations of other non-anadromous salmonids (Propst et al. 1992; Hepworth et al. 1997) may be partially explained by their choice of warmer, low-altitude streams.

*Source of Translocated Fish*

It does not appear that the source of translocated fish contributed to the failure of the 10 translocations thought to be affected by unsuitable habitat because early translocations with wild greenback cutthroat trout were about as successful as later ones

with hatchery fry (Table 1.1). Wild, adult fish were used only in eight of the earliest translocations of greenback cutthroat trout (during 1965-1981; three were later invaded by brook trout.) Hatchery fry and juveniles were used in the remaining translocations and to supplement most of the previous populations. Greenback cutthroat trout recovery has relied on hatchery broodstocks to found most new populations because it was believed that large numbers of fish were needed to rapidly establish new populations and that removal of wild fish would threaten wild populations (USFWS 1998).

Although our analysis suggests that translocating wild fish was not more successful at establishing populations than hatchery fry, sample sizes were small (Table 1.1). Translocations of wild individuals have been successful in native trout restoration (e.g., Hepworth et al. 1997) and have been more successful than releasing captive-reared birds and mammals worldwide (Griffith et al. 1989). Other reasons to use wild fish are to avoid the myriad of genetic and disease problems (e.g., whirling disease) associated with captive breeding (Dwyer and Rosenlund 1988; Philippart 1995; Snyder et al. 1996; Cunningham and Daszak 1998), and to replicate pure remnant populations to conserve genetic traits. Populations may be evolutionarily important if they have unique adaptations to marginal habitats (Scudder 1989), such as rapid egg development despite cold water temperatures reported for one remnant population (Dwyer and Rosenlund 1988). In contrast, a criticism of translocations of wild individuals is that too few fish are used to represent genetic diversity in the original population (Stockwell et al. 1996), but additions of relatively few fish in later years could remedy this (Mills and Allendorf 1996).

*Conclusions*

During the last 25 years, managers have successfully established 14 greenback cutthroat trout populations from translocations of wild and hatchery fish. Comparing these translocations to the 23 that failed suggests that the greenback cutthroat trout recovery program has been most successful in establishing populations in waters that were isolated from nonnative salmonids by natural barriers, had effective chemical treatments not impeded by complex habitats, previously supported reproducing trout populations, and had at least 2 ha of habitat. These results can help guide future translocation efforts for non-anadromous salmonids, but as previously noted, translocation waters were not chosen randomly so other unmeasured factors may contribute to success. Furthermore, judging the short-term (5-25 years) success of a translocation based on the recovery plan criteria is not the same as assessing long-term population persistence. It was assumed that these criteria would result in persistent populations, but confirming this will require data on population abundance, and size and age structure through time (Rieman and McIntyre 1993) that are not now available.

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## REFERENCES

Agresti, A. 1996. An introduction to categorical data analysis. John Wiley and Sons, Inc., New York.

Behnke, R. J. 1992. Native trout of western North America. American Fisheries Society, Monograph 6, Bethesda, Maryland.

Behnke, R.J., and M. Zarn. 1976. Biology and management of threatened and endangered Western trouts. U.S. Forest Service General Technical Report RM-28, Fort Collins, Colorado.

Chart, T.E., S. Gillihan, and D.A. Krieger. 1986. Greenback cutthroat trout recovery project. Colorado Division of Wildlife, Department of Natural Resources Progress Report 1986, Denver, Colorado.

Chart, T.E., R. Haneke, and K. Kehmeier. 1987. Greenback cutthroat trout recovery project. Colorado Division of Wildlife, Department of Natural Resources Progress Report 1987, Denver, Colorado.

Culver, S.R., and L.D. Lentsch. 1982. Greenback cutthroat trout recovery project. Colorado Division of Wildlife, Department of Natural Resources Progress Report 1981, Denver, Colorado.

Culver, S.R., and K.R. Bestgen. 1983. Greenback cutthroat trout recovery project. Colorado Division of Wildlife, Department of Natural Resources Progress Report 1982, Denver, Colorado.

Culver, S.R. 1983. Greenback cutthroat trout recovery project. Colorado Division of Wildlife, Department of Natural Resources Progress Report 1983, Denver, Colorado.

Culver, S.R, and K.R. Bestgen. 1985. Greenback cutthroat trout recovery project. Colorado Division of Wildlife, Department of Natural Resources Progress Report 1985, Denver, Colorado.

Cunjak, R. A., and G. Power. 1987. The feeding and energetics of stream-resident trout in winter. Journal of Fish Biology 31:493-511.

Cunningham, A. A., and P. Daszak. 1998. Extinction of a species of land snail due to infection with a microsporidian parasite. Conservation Biology 12:1139-1141.

Davis, J.A., and S. R. Culver. 1984. Greenback cutthroat trout recovery project. Colorado Division of Wildlife, Department of Natural Resources Progress Report 1984, Denver, Colorado.

Dwyer, W.P., and B.D. Rosenlund. 1988. Role of fish culture in reestablishment of greenback cutthroat trout. American Fisheries Society Symposium 4:75-80.

Dwyer, W.P., C.E. Smith, and B.D. Rosenlund. 1994. Recovery of the greenback cutthroat trout, a success story. Wild Trout Symposium Proceedings 5:174-176.

Foin, T. C., S. P. D. Riley, A. L. Pawley, D. R. Ayres, T. M. Carlsen, P. J. Hodum, and P. V. Switzer. 1998. Improving recovery planning for threatened and endangered species. BioScience 48:177-184.

Green, W. S. 1937. Colorado trout. Colorado Museum of Natural History Popular Series No. 2, Denver, Colorado.

Gresswell, R.E. 1988. Status and management of interior stocks of cutthroat trout. American Fisheries Society, Symposium 4, Bethesda, Maryland.

Griffith, B., J.M. Scott, J.W. Carpenter, and C. Reed. 1989. Translocation as a species conservation tool: status and strategy. Science 245:477-480.

Harig, A.L., and M.B. Bain. 1998. Defining and restoring biological integrity in wilderness lakes. Ecological Applications 8:71-87.

Hepworth, D. K., M. J. Ottenbacher, and L. N. Berg. 1997. Distribution and abundance of native Bonneville cutthroat trout (*Oncorhynchus clarki utah*) in southwestern Utah. Great Basin Naturalist 57:11-20.

Hickman, T.J., and D.L. Miller. 1977. A report on the greenback cutthroat trout inventory program, July to October 1977. Report of Colorado State University to Colorado Division of Wildlife, Denver, Colorado.

Hubert, W. A., R. W. Stonecypher, W. A. Gern, and J. Bobbit. 1994. Response of cutthroat trout embryos to reduced incubation temperatures at different developmental stages. The Progressive Fish-Culturist 56:185-187.

Hunt, R. L. 1969. Overwinter survival of wild fingerling brook trout in Lawrence Creek, Wisconsin. Journal of the Fisheries Research Board of Canada 26:1473-1483.

Jordan, D.S. 1891. Report of explorations in Colorado and Utah during the summer of 1889, with an account of the fishes found in each of the river basins examined. Bulletin of the United States Fish Commission 9:1-40.

Kehmeier, K., R. Haneke, and T. Garrison. 1988. Greenback cutthroat trout recovery project. Colorado Division of Wildlife, Department of Natural Resources Progress Report 1988, Denver, Colorado.

Kehmeier, K., and R. VanBuren. 1990. Greenback cutthroat trout recovery project. Colorado Division of Wildlife, Department of Natural Resources Progress Report 1989-1990, Denver, Colorado.

Langlois, D., T. Hickman, and D. Miller. 1978. Greenback cutthroat trout restoration program. Colorado Division of Wildlife, Endangered Wildlife Investigations Performance Report SE-3-1, Denver, Colorado.

Langlois, D., T. Hickman, and D. Miller. 1979. Greenback cutthroat trout restoration program. Colorado Division of Wildlife, Endangered Wildlife Investigations Progress Report SE-3-2, Denver, Colorado.

Lubow, B. C. 1996. Optimal translocation strategies for enhancing stochastic metapopulation viability. Ecological Applications 6:1268-1280.

MacPhee, C. 1966. Influence of differential angling mortality and stream gradient on fish abundance in a trout-sculpin biotone. Transactions of the American Fisheries Society 995:381-387.

Mills, L.S., and F.W. Allendorf. 1996. The one-migrant-per-generation rule in conservation and management. Conservation Biology 10:1509-1518.

Mullan, J.W. 1973. Considerations in perpetuation of greenback cutthroat trout (*Salmo clarki stomias*). U.S. Fish and Wildlife, Rocky Mountain National Park Special Report, Estes Park, Colorado.

Philippart, J.C. 1995. Is captive breeding an effective solution for the preservation of endemic species? Biological Conservation 72:281-295.

Propst, D. L., J. A. Stefferud, and P. R. Turner. 1992. Conservation and status of Gila trout, *Oncorhynchus gilae*. Southwestern Naturalist 37:117-125.

Rieman, B. H., and J. D. McIntyre. 1993. Demographic and habitat requirements for conservation of bull trout. U.S. Forest Service General Technical Report INT-302.

Rinne, J. N. 1980. Spawning habitat and behavior of Gila trout, a rare salmonid of the southwestern United States. Transactions of the American Fisheries Society 109:83-91.

Rinne, J.N., W.L. Minckley, and J.N. Hanson. 1981. Chemical treatment of Ord Creek, Apache County, Arizona, to re-establish Arizona trout. Journal of the Arizona-New Mexico Academy of Science 16:74-78.

Rosenlund, B.D., K.M. Canestorp, D.E. Sharps, and J. Barrett. 1994. Fisheries and aquatic management, Rocky Mountain National Park 1993. U.S. Fish and Wildlife Service, Colorado Fish and Wildlife Assistance Office Report, Golden, Colorado.

Rosenlund, B.D., and D.R. Stevens. 1988. Fisheries and aquatic management, Rocky Mountain National Park 1987. U.S. Fish and Wildlife Service, Colorado Fish and Wildlife Assistance Office Report, Golden, Colorado.

Rosenlund, B.D., and D.R. Stevens. 1990. Fisheries and aquatic management, Rocky Mountain National Park 1988-1989. U.S. Fish and Wildlife Service, Colorado Fish and Wildlife Assistance Office Report, Golden, Colorado.

Scudder, G.G.E. 1989. The adaptive significance of marginal populations: a general perspective. Pages 180-185 *in* C.D. Levings, L.B. Holtby, and M.A. Henderson, editors. Proceedings on the national workshop on effects of habitat alteration on salmonid stocks. Canadian Special Publication of Fisheries and Aquatic Sciences 105.

Shuter, B.J., and J.R. Post. 1990. Climate, population viability, and the zoogeography of temperate fishes. Transactions of the American Fisheries Society 119:314-336.

Simons, L. H., D. A. Hendrickson, and D. Papoulias. 1989. Recovery of the Gila topminnow: a success story? Conservation Biology 3:11-15.

Smith, K., and M.E. Lavis. 1975. Environmental influences on the temperature of a small upland stream. Oikos 26:228-236.

Snyder, N.F.R., S.R. Derrickson, S.R. Beissinger, J.W. Wiley, T.B. Smith, W.D. Toone, and B. Miller. 1996. Limitations of captive breeding in endangered species recovery. Conservation Biology 10:338-348.

S-PLUS 4. 1997. S-PLUS 4 guide to statistics. Data Analysis Products Division, MathSoft, Seattle, Washington.

Stevens, D.R., and B.D. Rosenlund. 1986. Greenback cutthroat trout restoration in Rocky Mountain National Park. Pages 104-118 *in* G. Larson and M. Soakup, editors. Conference on Science in the National Parks, Colorado State University, Fort Collins, Colorado.

Stockwell, C.A., M. Mulvey, and G.L. Vinyard. 1996. Translocations and the preservation of allelic diversity. Conservation Biology 10:1133-1141.

Stonecypher, R. W., W. A. Hubert and W. A. Gern. 1994. Effect of reduced incubation temperature on survival of trout embryos. The Progressive Fish-Culturist 56:180-184.

Stuber, R.J., B.D. Rosenlund, and J.R. Bennett. 1988. Greenback cutthroat trout recovery program: management overview. American Fisheries Society Symposium 4:71-74.

Thurow, R. T., and J. G. King. 1994. Attributes of Yellowstone cutthroat trout redds in a tributary of the Snake River, Idaho. Transactions of the American Fisheries Society 123:37-50.

Tyler, T., W. J. Liss, L. M. Ganio, G. L. Larson, R. Hoffman, E. Deimling, and G. Lomnicky. 1998. Interaction between introduced trout and larval salamanders (*Ambystoma macrodactylum*) in high-elevation lakes. Conservation Biology 12:94-105.

USFWS (U.S. Fish and Wildlife Service). 1998. Greenback cutthroat trout recovery plan. U. S. Fish and Wildlife Service, Denver, Colorado.

Wang, L., and R.J. White. 1994. Competition between wild brown trout and hatchery greenback cutthroat trout of largely wild parentage. North American Journal of Fisheries Management 14:475-487.

Washington, T. 1981. Greenback cutthroat trout restoration program. Colorado Division of Wildlife, Endangered Wildlife Investigations Progress Report SE-3-3, Denver, Colorado.

Williams, J. E., and 12 coauthors. 1988. American Fisheries Society guidelines for introductions of threatened and endangered fishes. Fisheries 13(5):5-11.

Wiltzius, W.J. 1985. Fish culture and stocking in Colorado, 1872-1978. Colorado Division of Wildlife, Report DOW-R-D-12-85, Fort Collins, Colorado.

Winters, D.S., E.N. Bennett, and J.P. Gallagher. 1995. Basinwide stream habitat inventory: greenback cutthroat trout streams of the Pike and San Isabel National Forests in Colorado. U.S. Forest Service, Pike and San Isabel National Forests Report, Rye, Colorado.

Young, M.K. 1995. Conservation assessment for inland cutthroat trout. U.S. Forest Service General Technical Report RM-GTR-256, Fort Collins, Colorado.

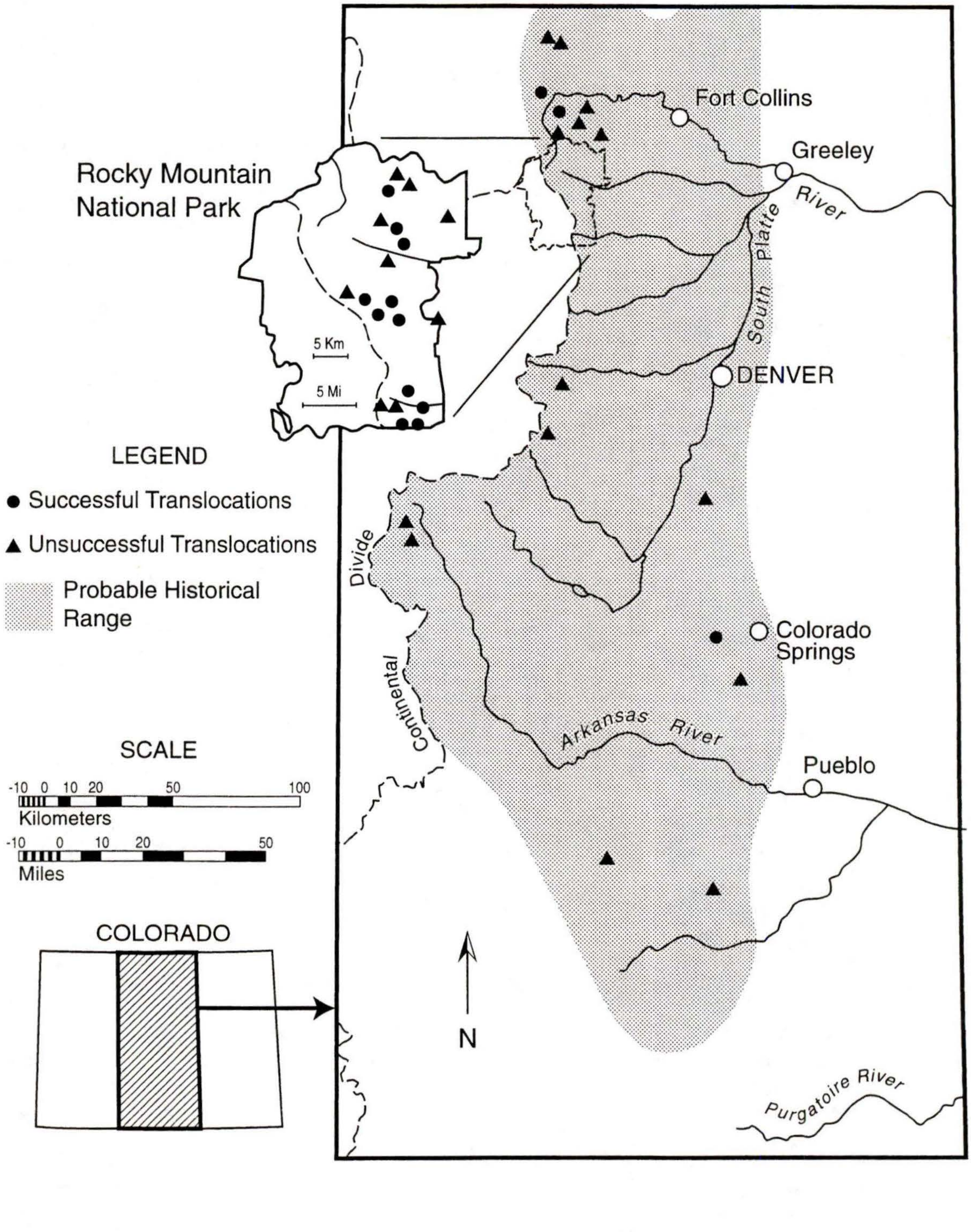
Table 1.1. Success of 24 greenback cutthroat trout translocations as functions of habitat area, previous ability of habitat to support trout, median elevation, and source of translocated fish. Eleven populations subsequently invaded by brook trout and two depressed by other factors were excluded (see footnotes).

| Variable | Number of translocations | Percent successful | Fisher's exact test (*P*-value) |
|---|---|---|---|
| Habitat area of lakes and streams[a] | | | |
| <2 ha | 10 | 30% | 0.035 |
| ≥2 ha | 14 | 79% | |
| Ability to support trout[b] | | | |
| Previously barren | 10 | 40% | 0.211[b] |
| Supported trout | 14 | 71% | |
| Median elevation | | | |
| <3000 m | 9 | 56% | 1.000 |
| ≥3000 m | 15 | 60% | |
| Source of translocated fish | | | |
| Wild adults | 5 | 40% | 0.615 |
| Hatchery fry and juveniles | 19 | 63% | |

[a] Total habitat area of stream segments was calculated using mean width (3.5 m) from field surveys of 10 streams where greenback cutthroat trout were translocated (Harig and Fausch, unpublished).

[b] Eleven other habitats supported reproducing populations of brook trout and may have supported greenback cutthroat trout prior to reinvasion, yielding a total success rate of 84% ($P = 0.016$ by Fisher's exact test, n=35).

Figure 1.1. The distribution in 1999 of translocated populations for which sufficient time was available to assess population status (n=37; see text). Translocations were considered successful (n=14) if they met recovery plan criteria and unsuccessful (n=23) if they were depressed by nonnative salmonids, unsuitable habitat, or other factors (bird predation, mining pollution).



Fort Collins
Greeley
River
South Platte
DENVER
Rocky Mountain
National Park
5 Km
5 Mi
LEGEND
● Successful Translocations
▲ Unsuccessful Translocations
Probable Historical Range
Divide
Continental
Colorado Springs
Arkansas River
Pueblo
SCALE
-10 0 10 20 50 100
Kilometers
-10 0 10 20 50
Miles
COLORADO
N
Purgatoire River

# CHAPTER II:

# MINIMUM HABITAT REQUIREMENTS FOR ESTABLISHING TRANSLOCATED CUTTHROAT TROUT POPULATIONS

## ABSTRACT

Translocation is an important management strategy in a majority of conservation programs for endangered or threatened species, including native cutthroat trout (*Oncorhynchus clarki*) in the western United States. Most subspecies of cutthroat trout have declined to less than 5% of their historical range, and both historical and translocated populations now persist in small isolated fragments of habitat. Translocations remain one of the few methods available for increasing their range, but success rates are generally less than 50%, and habitat quality or quantity are frequently cited as the cause of failure. Therefore, managers would benefit from a quantitative assessment of specific habitat attributes that contribute to the success or failure of translocations.

We conducted field surveys of stream-scale habitat and measured basin-scale habitat attributes using a Geographic Information System for 27 streams where two subspecies of cutthroat trout were translocated in their native range in Colorado and New Mexico. We developed habitat models using polytomous logistic regression that

compare attributes of sites where translocations established relatively high numbers of cutthroat trout, low numbers of cutthroat trout, and few or no cutthroat trout (absent), to identify macrohabitat features that promote establishment and persistence of native cutthroat trout populations. Models of stream-scale habitat attributes indicated that cold summer water temperature, narrow stream width, and lack of deep pools limit populations of cutthroat trout. Cold summer temperatures are known to delay spawning and prolong egg incubation, which reduces the growth of fry and likely limits their overwinter survival. Furthermore, small streams with few deep pools may lack the space necessary to permit overwinter survival of a sufficient number of individuals to sustain a viable population. Models of basin-scale habitat were not as effective as stream-scale habitat models for distinguishing among translocation sites with high, low, or absent population status, but indicate that watershed area is useful as a coarse filter for separating sites with high numbers of cutthroat trout from those with low or absent status. Large watersheds (> 14.7 km$^2$) are expected to encompass low elevation habitat that provides warmer summer temperatures, and have relatively wide stream channels of sufficient length to provide an adequate number of deep pools. These macrohabitat models will be valuable to managers for selecting future translocation sites with a high probability of success and for identifying whether populations in fragments of historical habitat are likely to persist.

## INTRODUCTION

Translocation of individuals to establish, reestablish, or supplement a population is often an important management strategy in the conservation of endangered or

threatened animals (Griffith et al. 1989). In a review of recovery plans for threatened or endangered species, 70% of all recovery programs (Tear et al. 1993), and over 80% of programs for fish (Williams et al. 1988), called for translocations. Some highly publicized translocation programs, many of which successfully founded self-sustaining populations (e.g., American bison [*Bison bison*], Kleiman 1989; Peregrine falcon [*Falco peregrinus*], Enderson et al. 1995, Millsap et al. 1998; see also Wolf et al. 1996), have established translocation as an effective management tool to natural resource managers and the general public. However, success rates for translocations of birds, mammals, and fish are generally less than 50% (Williams et al. 1988; Griffith et al. 1989; Simons et al. 1989; Hendrickson and Brooks 1991; Harig et al. in review). A survey of bird and mammal translocations indicates that habitat quality of the translocation site, number of individuals released, and the proximity of the site to the core of the species' historical distribution are three main factors influencing success (Griffith et al. 1989; Wolf et al. 1996, 1998). While these general patterns are useful for identifying areas where research is needed, they do not provide specific information for selecting a translocation site with a high probability of success. For example, the number of individuals needed to establish a self-sustaining population and the factors defining quality habitat may be specific to the taxon being considered for translocation. Unfortunately, most translocations have been inadequately documented, with much of the information unpublished or in reports with limited access, and few have had sufficient population monitoring to identify these factors (Minckley 1995; Hodder and Bullock 1997). Therefore, there is a need for quantitative assessment of specific ecological factors that contribute to the success or failure of translocations.

Native subspecies of cutthroat trout (*Oncorhynchus clarki*) in the western United States have been reduced to a small portion of their historical range, nearly all less than 5%, primarily due to habitat degradation and interactions with nonnative salmonids (Gresswell 1988; Behnke 1992; Young 1995a). Of the 14 subspecies recognized (Behnke 1992; three are undescribed), one is extinct, four are listed as threatened or endangered under the Endangered Species Act, and conservation plans have been developed for most others. Establishing new cutthroat trout populations through translocation of genetically pure trout into fishless waters or those treated with toxicants to remove nonnative salmonids remains one of the few management strategies for increasing their range (Stuber et al. 1988; Young 1995b; USFWS 1998). Analysis of cutthroat trout translocations to identify factors that promote establishment and persistence of populations may improve the success of future translocations and provide an example for assessment of other recovery programs. Of 37 attempts to establish allopatric populations of greenback cutthroat trout (*O. c. stomias*), only 38% were successful, whereas 30% were reinvaded by nonnative salmonids, 27% apparently had unsuitable habitat, and 5% were depressed by other factors (Harig et al. in review). Similarly, 46% of 28 Rio Grande cutthroat trout (*O. c. virginalis*) translocations established naturally reproducing populations, whereas 32% were reinvaded, and 21% had unsuitable habitat (Harig and Fausch 1996 and unpublished data; Alves 1998; Bill Stumpff, New Mexico Department of Game and Fish [NMDGF], personal communication). Cutthroat trout readily hybridize with other spring-spawning salmonids like rainbow trout (*O. mykiss*), and are apparently displaced by fall-spawning species such as brook trout (*Salvelinus fontinalis*) and brown trout (*Salmo trutta*; Wang and

White 1994), so reinvasion by nonnative salmonids results in a translocation failure. Isolating barriers, while protecting some cutthroat trout populations from upstream migration of nonnative salmonids, restrict them to areas that may be too small or have insufficient habitat to support a viable population (Moyle and Sato 1991; Moyle and Yoshiyama 1994).

Habitat quality has been emphasized as an essential component of translocation success (IUCN 1987, 1995; Williams et al. 1988; Kleiman 1989; Moyle and Sato 1991; Gordon 1994; Minckley 1995) and was the factor most often cited as leading to translocation failure (Griffith et al. 1989; Wolf et al. 1996, 1998). Success requires that habitat is of sufficient quality to meet the life history requirements of the species (Williams et al. 1988) and large enough to support a population that is self-sustaining in the face of demographic and environmental stochasticity (Moyle and Sato 1991). Some conservation programs for mammals have recommended against using translocations because sufficient habitat is not available (Ruth et al. 1998; Struhsaker and Siex 1998). Research on the minimum habitat requirements of a species may be necessary to identify a suitable translocation site, particularly if those contributing to translocation failure are not apparent (Hodder and Bullock 1997). Some species show metapopulation dynamics or use several spatially segregated sites with different resources in order to persist, so measuring regional spatial dynamics may be needed to identify their habitat requirements (Hodder and Bullock 1997).

There are few empirical data on minimum habitat requirements for entire salmonid populations because ecologists and fish biologists have historically focused on fine spatial scales to understand how environmental factors influence local abundance

and dynamics (cf. Gowan et al. 1994; Fausch and Young 1995; Schlosser and Angermeier 1995). High levels of natural spatial and temporal variation among sites at fine spatial scales can mask factors that influence stream fish populations (Hicks et al. 1991; Dunham and Vinyard 1997; Lohr and Fausch 1997), resulting in habitat models that lack generality for application beyond the stream or watershed for which they were developed (Fausch et al. 1988; Poff and Ward 1989; Rieman and McIntyre 1995). Moreover, fine spatial scales may not adequately reflect the life history of the species being studied (Torgersen et al. 1999; Labbe and Fausch in press). Salmonids and other stream fishes vary markedly in habitat use patterns during their life cycle, requiring appropriate temperatures, flow, substrate, and physical structure at each life history stage (Bisson et al. 1981; Schlosser 1995). These different habitats may be separated (Schlosser 1991, 1995), especially in disturbed watersheds (Fausch et al. 1995), so high spatial heterogeneity and connectivity of habitat patches is needed to maintain persistent salmonid populations. Therefore, analyses that include coarse-scale processes are likely to be most appropriate for stream populations of cutthroat trout.

The goal of our research was to identify macrohabitat features that promote establishment and persistence of translocated native cutthroat trout populations isolated in headwater streams by fish movement barriers. For study, we chose two subspecies of cutthroat trout in Colorado and New Mexico with nearly identical ecological requirements. We compared habitat attributes at stream and basin scales in streams where translocations successfully established a naturally reproducing cutthroat trout population to those where translocated populations were extirpated for reasons apparently related to habitat size or quality rather than invasion by nonnative salmonids. We fit

models that predict risk of translocation success or failure from macrohabitat measured at these two scales, which also provide information about habitat factors that are likely critical for native cutthroat trout isolated in headwater streams. They will be valuable to managers for choosing future restoration sites with a high probability of establishing a successful cutthroat trout population through translocation and for identifying whether populations in fragments of historical habitat are likely to persist.

## SALMONID HABITAT AND POTENTIAL LIMITING FACTORS

### *Stream-Scale Habitat*

Based on a literature search of stream salmonid ecology, we made a series of *a priori* predictions about the factors that potentially limit translocated cutthroat trout populations. Translocation sites are generally headwater reaches in high elevation streams isolated by barriers to fish movement, which protects the cutthroat trout population from invasion by nonnative salmonids but restricts them to a small stream segment that may not provide the minimum habitat necessary for population persistence (Moyle and Sato 1991; Moyle and Yoshiyama 1994). Persistent salmonid populations require enough habitat to support a sufficient number of adults to sustain a population, refuge from high flow during spring snowmelt run-off, refuge from low temperature and low flow during winter, optimum summer temperatures to promote spawning, incubation, and emergence prior to the onset of winter, and sufficient clean (i.e., silt-free) gravel to construct spawning redds (Bisson et al. 1981; Behnke 1992). Therefore, we predicted that cutthroat trout persistence is less likely in streams having short length, few pools,

small or shallow pools, pools with little physical structure providing refuges from flow, low winter and summer water temperatures, and little clean spawning gravel.

Habitat for cutthroat trout must provide sufficient space of appropriate quality to support a naturally reproducing population that is large enough to withstand the extirpation risks from environmental catastrophes and demographic stochasticity (Moyle and Sato 1991). In summer, adult salmonids favor pools created by large woody debris, boulders, or lateral scour (Bisson et al. 1981; Flebbe and Dolloff 1995; Young 1996), probably because they provide cover through high structural habitat complexity (Fausch and Northcote 1992; Richmond and Fausch 1995) and abundant food through invertebrate habitat (Angermeier and Karr 1984; Benke et al. 1985). Juvenile salmonids, which are susceptible to displacement by high water velocities (Heggenes and Traaen 1988; Nehring and Anderson 1993; Latterell et al. 1998), tend to occupy run and pool margins (Bisson et al. 1981) and backwater pool habitats characterized by low water velocities, abundant detritus, and abundant invertebrates (Moore and Gregory 1988). Therefore, short stream segments with only small, shallow, structurally simple pools may not provide the minimum habitat necessary to support high numbers of cutthroat trout.

In winter, survival depends on having adequate refugia from low temperature and low flow. Salmonids tend to aggregate in deep pools with low flow velocities and areas of cover (Bustard and Narver 1975; Chisholm et al. 1987; Griffith and Smith 1993) or near sources of groundwater discharge (Cunjak and Power 1986). Thick surface ice resulting from severe winter temperatures threatens salmonid survival if few deep pools are present (Chisholm et al. 1987). Cutthroat trout at high elevations may also be subject to rapid winter temperature acclimation and overwinter starvation, so survival depends on

their ability to attain a body size large enough to withstand metabolic deficits (Hunt 1969; Cunjak and Power 1987). This is particularly important for young cutthroat trout because most metabolic functions are limited by body size (Shuter and Post 1990). Therefore, streams with few deep pools and low winter water temperatures may not be able to support overwintering cutthroat trout.

Cutthroat trout are a spring-spawning species with specific temperature and substrate requirements. Water temperatures below a daily maximum of 4 to 8 °C can delay spawning (Rinne 1980; Thurow and King 1994) and prolong egg incubation, lowering embryo survival and increasing time to hatching (Hubert et al. 1994; Stonecypher et al. 1994). Embryos that hatch late may not be able to attain a body size needed to survive the winter energy deficit (Hunt 1969; Cunjak and Power 1987). Clean, silt-free gravel, predominantly less than 4 cm in diameter, is used for egg deposition (Rinne 1980; Thurow and King 1994). A high proportion of fine sediment in a redd can lead to low fry emergence success, possibly from lack of dissolved oxygen (Weaver and Fraley 1993), which may ultimately reduce juvenile recruitment and influence adult population levels (Scrivener and Brownlee 1989; Beard and Carline 1991). Therefore, streams with low summer temperatures and insufficient clean gravel may not be able to support a naturally reproducing cutthroat trout population.

### *Basin-Scale Habitat*

Stream ecosystems may be viewed as hierarchically organized physical environments consisting of microhabitat, habitat, stream, subbasin, and drainage basin units (Frissell et al. 1986). Each level in the physical hierarchy is controlled by processes

functioning at particular temporal and spatial scales that are nested within the next level of organization. Theoretically, the functional processes that structure the coarse-scale levels of organization i.e., subbasin and drainage basin, also influence habitat at finer spatial scales (Platts 1979; Frissell et al. 1986; Lanka et al. 1987; Poff 1997). For example, watershed area influences the amount of sediment input into the channel and total discharge, slope regulates flow velocity and pool morphology, and aspect affects vegetation and stream temperature (Mark 1975; Dunne and Leopold 1978; Hicks et al. 1991; Moore et al. 1991; Eash 1994). Thus, coarse-scale analyses of attributes that govern vegetative patterns, drainage networks, erosive mechanisms, and fluvial processes (Lanka et al. 1987; Overton et al. 1995) should be useful for predicting the potential habitat quality for stream salmonids (Poff and Ward 1989; Nelson et al. 1992; Rieman and McIntyre 1995; Schlosser 1995; Schlosser and Angermeier 1995).

Based on this theory, we predicted that cutthroat trout persistence is less likely in sites with small watershed area, short channel length, high elevation, high basin relief, high latitude, steep channel slope, and a North-South aspect. Small watersheds and short channels may not be able to support a population large enough to withstand possible extirpation from environmental and demographic stochasticity or catastrophic events (Lande 1993). Translocation sites at high elevation and high latitude or with a North-South aspect may have colder temperature regimes that cannot support overwintering trout or successful reproduction. Finally, streams with high channel relief and steep channel slope may not have sufficient refuge from high flow or enough low gradient pool habitat to support trout, especially juveniles.

## METHODS

### *Study Sites*

We selected 27 cutthroat trout translocation streams (12 greenback and 15 Rio Grande) in Colorado and New Mexico for study (Figure 2.1; Table 2.1). These sites represent all but two known stream translocations not invaded by nonnative salmonids of these cutthroat trout subspecies made throughout their historic range prior to 1996 (Harig and Fausch 1996; Harig et al. in review; B. Stumpff, personal communication). We also included one additional stream where a translocation was conducted in 1997 to demonstrate use of the final model (i.e., as a test site). Other translocations conducted during 1996 and after were excluded because they were too recent to assess population status. Translocations into lakes alone were also excluded because lake habitat is not directly comparable to stream habitat.

Before translocation of wild or hatchery cutthroat trout, streams were either barren of fish or had been chemically treated with a fish toxicant (antimycin or rotenone) to remove nonnative salmonids. Unpublished data from natural resource agencies on the frequency, number, size, and source (hatchery-reared, wild broodstock, or wild) of translocated cutthroat trout were incomplete (see Appendix II), but do not suggest that initial stocking practices influenced success of most cutthroat trout translocations. In some streams, cutthroat trout populations were established from introductions of relatively few wild fish (e.g., 182 for San Francisco Creek), whereas others failed to support high numbers of cutthroat trout despite repeated stocking of both hatchery and wild individuals (e.g., five stocking events totaling > 6200 in May Creek). Typically, Rio Grande cutthroat trout populations were founded from one to three translocations

averaging 30 to 1000 wild fish ranging in size from fry to small adults (15 cm). Most greenback cutthroat trout populations were founded from two to five introductions of 400 - 5000 hatchery fry and juveniles (< 8 cm), although small adults were included in some introductions.

All translocation streams were headwater sites above 2400 m in elevation (Table 2.1) and their headwaters extended to elevations as high as 3600 m. Stream habitat generally alternated between steep, forested reaches with conifers in riparian zones, and low-gradient, meandering reaches lined with forbes, willows (*Salix* spp.), or cottonwoods (*Populus* spp.). Mean channel gradients, measured from digital data using a Geographic Information System (GIS), ranged from 7.3 to 20.2% (Table 2.1), but these are likely to be higher than actual gradients due to greater stream sinuosity (and therefore greater channel length) on the ground than can be shown on maps. Length of summer habitat for cutthroat trout ranged from 1.0 to 20.5 km (Table 2.1), and was isolated from encroachment of nonnative salmonids by barriers to upstream fish movement including waterfalls, cascades, steep gradients, dry channels, or manmade structures like rock-filled gabions (Table 2.1). Nine streams (Cony, Fern, Greenhorn, Jacks, Medano, Ouzel, and West creeks, Roaring River, and Rough Canyon/Rhodes Gulch) also have waterfalls or steep cascades in their middle reaches that divide them into multiple habitats. We considered them single populations even though fisheries managers regard four of them as multiple populations because downstream reaches potentially depend on emigration of cutthroat trout from upstream areas.

*Field Surveys of Stream-Scale Habitat and Cutthroat Trout Abundance*

We conducted field surveys of stream-scale habitat and cutthroat trout abundance along the entire length of each translocation stream from the downstream fish movement barrier upstream to the end of pool habitat, where the bankfull channel width was < 2.0 m and the wetted width was usually ≤ 1.0 m. Surveys were conducted during June through October 1996 through 1998. During each of the first two years, we randomly selected streams from among 29 sites in Colorado where either cutthroat trout subspecies had been translocated and managers reported that nonnative salmonids had not invaded. However, we found that nonnative brook trout had invaded six of them, so in 1998 we randomly selected five additional translocation streams (of seven known; B. Stumpff, personal communication) from the rest of the range of Rio Grande cutthroat trout in New Mexico. One of these later proved to have been translocated after 1995, and so is used as a test of the final model.

We counted the number of fish observed in pool and fast-water channel units (i.e., riffle, run, or cascade; Hawkins et al. 1993) to determine: (1) which reaches were inhabited by trout in the summer, (2) if they had successfully reproduced, based on observing age-0 or age-1 trout, and (3) the minimum abundance of trout. These visual fish counts were not intended as population estimates, but as measures of the minimum number of trout supported by the stream for classifying relative translocation success. The number, species, and approximate size of each fish were determined using polarized glasses by carefully approaching a channel unit from downstream. Cutthroat trout generally hold positions in the open water near the surface (Griffith 1972; Nakano et al. 1998), so are highly visible. However, we used a depth staff to sweep beneath undercut

banks, large woody debris, and boulders to detect any additional fish. Usually, none were found. We also recorded water transparency during surveys to help assess whether the fish count was hindered by turbidity, turbulence, deep water, low light levels, or thick vegetation, and thereby underestimated relative to other streams. However, most streams were small and clear, so cutthroat trout were easily observed if approached quietly.

We surveyed stream-scale habitat at each pool, defined as a channel unit that was at least a ½ channel width long, relatively deep (minimum residual depth of 18 cm) and slow flowing, with a gentle surface water slope (criteria adapted from Hawkins et al. 1993). Stream surveys focused on pool habitat and the structure and spawning gravel they provided because salmonids primarily rely on pools for both summer and winter habitat. We measured bankfull pool width, residual depth, presence of sediment-free substrate, and physical habitat structure for trout (Table 2.2). The number of variables we could include in a model of translocation success was limited by the modest sample size of streams, so we calculated the geometric mean of the number of pools with large woody debris, boulders, and $\geq 0.2$ m of undercut bank to create one variable indicating the number of pools with physical structure.

Stream temperatures were measured ($\pm 0.2$ °C) for each translocation stream at least every 96 minutes for a minimum of one year using either an Optic StowAway® or TidbiT® thermograph (Onset Computer Corporation, Pocasset, MA) placed in the largest pool. Thermographs were placed in each stream when first surveyed and replaced each year. Thermal regimes were measured for three years (1996-1999) in 8 streams (i.e., thermographs were first installed in 1996), at least two years in 16 streams, and at least one year in all 28 streams. We analyzed two thermal characteristics; mean daily

temperature for the months of June through August to encompass periods of egg incubation and emergence of cutthroat trout fry, and for the 3-mo. period from December through February to measure mean overwinter temperatures. For four translocation streams that contained more than one thermograph (Medano Creek and tributaries, Rough Canyon/Rhodes Gulch, San Francisco Creek and tributaries, Cony Creek and Lower Hutcheson Lake), daily temperatures were averaged.

*Measurement of Basin-Scale Habitat using Digital Data*

A Geographic Information System and digital map data were used to measure basin-scale habitat attributes for the 28 greenback and Rio Grande cutthroat trout translocation streams surveyed in the field and all 70 known remnant, historical stream populations not invaded by nonnative salmonids (7 greenback and 63 Rio Grande). Digital elevation models (DEMs) corresponding to 1:24,000 scale topographic quadrangles were provided by the U.S. Forest Service (USFS) Geometronics Unit in Salt Lake City, UT (D. Wolf, USFS) or purchased from the U.S. Geological Survey (USGS). These 7.5-minute grids were in raster format with 30 m x 30 m resolution.

We used the hydrological modeling tools in the GRID® module of Arc/Info® (ESRI 1995; specific commands are listed in capitals letters) to quantify basin attributes (Table 2.3) from the DEMs using a series of functions that set up the proper DEM conditions and delineated basin boundaries and stream networks. To set up proper conditions, DEMs were converted to elevation grids (DEMLATTICE), sinks were filled using appropriate procedures in GRID® (FLOWDIRECTION, SINK, WATERSHED, ZONALFILL, ZONALMIN, FILL) and adjacent quadrangles were joined (MOSAIC).

Sinks, which are areas surrounded by higher elevations, are usually data errors resulting from the spatial interpolation procedure used to create the DEM and can create false stream networks. Lakes were also filled even though they are natural sinks because they were part of cutthroat trout stream habitat and were not modeled as separate basins.

Stream networks were identified from the depressionless grids using the commands FLOWDIRECTION to find the steepest descent from each cell, and FLOWACCUMULATION to calculate the number of upslope cells contributing flow to each cell. Cells with high flow accumulation were used to identify stream channels. We chose cells that accumulated flow from at least 1000 other cells (CON) to delineate streams because this value most closely matched the smallest flowing channels in field surveys. Cells with no flow accumulation were used to identify drainage basin boundaries. We set the downstream end of cutthroat trout basins as the fish movement barrier (SNAPPOUR) to delineate the watershed (WATERSHED). The other data layers, including elevation, stream network, slope (SLOPE), and aspect (ASPECT), were then cropped to calculate watershed area, channel length, latitude, and topographic relief that define characteristics of basin and channel morphology (Table 2.3).

*Statistical Models of Translocation Success*

We used the predictions described in *Salmonid Habitat and Potential Limiting Factors* to develop four sets of formal *a priori* statistical models predicting cutthroat trout translocation success; one from summer water temperatures, a second from water temperature and stream-scale habitat attributes measured during field surveys, a third from basin-scale habitat attributes measured using digital data, and a fourth from a

combination of the variables in the "best" models of the previous model sets. Each group of models is a nested set that represents formal hypotheses about attributes that potentially limit persistence of translocated cutthroat trout populations (Burnham and Anderson 1998). Summer water temperatures were analyzed as a separate model set to determine which month was the "best" measure to include in models of stream-scale habitat attributes. For this analysis, the 27 streams were assumed to be a random sample of all streams where managers might attempt translocations. Given this, models can be used to predict success of future translocations.

Response variables for the models were based on ranked categories of translocation success (absent, low, high) determined from our visual estimates of cutthroat trout minimum abundance. We assumed that streams supporting relatively *high* numbers of cutthroat trout had minimally sufficient habitat, streams supporting relatively *low* numbers had marginal habitat, and streams *absent* of cutthroat trout had insufficient habitat. Number of dependent variables per model was limited to three because too many variables can result in a statistically unstable model for small data sets (i.e., over-fitting; Burnham and Anderson 1998). Therefore, interaction and quadratic terms were excluded except for the interaction between latitude and elevation in models of basin-scale habitat. All pairwise correlations of independent variables were evaluated to assess multicollinearity.

All models were fit using ordinal polytomous logistic regression (PLR; Agresti 1996; SAS Institute Inc. 1996). Ordinal PLR directly incorporates ordering of response categories, which results in models with simpler interpretations and potentially greater power than ordinary multicategory logit models (Agresti 1996). Model diagnostics

included the Score Test for Proportional Odds Assumption to determine the validity of choosing ordinal over nominal PLR, and Deviance and Pearson goodness-of-fit statistics to examine model fit (SAS Institute Inc. 1996). Overall model significance was reported from the likelihood-ratio statistic for an ordinal test of independence ($H_o$: $\beta = 0$; Agresti 1996). For specific habitat models where the differences between predicted values for two or more status categories were similar, binary logistic regression was used for pairwise comparison of these subsets of the data.

One or more "best approximating" models were selected from each set of candidate models using Akaike's Information Criterion corrected for small-sample bias (AICc; Burnham and Anderson 1998). Models were ranked using AICc weights, which is a measure of the weight of evidence in favor of a model given the data. The model with the highest weight is considered the "best" model. However, if there is not one model that is clearly the best, models within two AICc points of the highest weighted model are considered competing models. Results from model-averaging of the competing models based on the AICc weights provides a more precise, stable inference than from only one "best" model (Burnham and Anderson 1998). Model selection based on an information–theoretic approach such as this is superior to traditional hypothesis testing for this dataset because it allows comparison of more than two models at once, balances precision and bias in selecting an appropriate model, and does not require that the data were collected from a formal designed experiment (Burnham and Anderson 1998). It has been successfully used in similar ecological research to define optimal habitat for northern spotted owl juvenile survival (Franklin et al. in press).

## RESULTS

### *Relative Population Status*

The 27 study streams to which cutthroat trout were translocated prior to 1996 were classified based on their population status to compare habitat variables and develop models of translocation success. Population status was assigned based on minimum trout abundance, the number of age-1 and older cutthroat trout observed during the visual survey (Table 2.4). If less than four age-1 and older trout were observed, then it was assumed that the stream was unable to support a population of cutthroat trout, and it was assigned a rating of *absent*. In none of these streams did other factors like turbidity reduce visibility of trout. Streams where less than 100 trout were observed supported a population with relatively low numbers of cutthroat trout and were rated *low*. Streams with more than 100 cutthroat trout were rated *high*. We chose the criterion of 100 trout to separate low versus high populations because streams with counts between 100 and 200 trout and one with a count of 61 (Fern Creek) were likely underestimates relative to other surveys. All but one of these streams had lakes or large beaver ponds, habitats where it was not possible to make accurate visual counts. All other streams with lakes or large beaver ponds had counts greater than 200 cutthroat trout and were ranked as high, so we felt it appropriate to rank the streams with counts between 100 and 200 as high also.

To assess the accuracy of our population status categories, we compared our basin-wide visual counts and subsequent status rating to estimates of standing stock (kg/ha) made by two-pass removal electrofishing (cf. Otis et al. 1978) for shorter reaches of 22 streams for which data were available from natural resource management agencies. Our visual estimates, calculated as minimum density of trout (no./km), were positively

correlated with agency estimates of standing stock (kg/ha; $r = 0.70$, $P = 0.003$, $n = 22$), and in 21 of 22 cases yielded similar status ratings to those designated by fisheries managers (Alves 1998; USFWS 1998). The differences in trout abundance estimated by the two methods for individual streams may be attributed primarily to differences in the length and habitat of reaches sampled. Our visual sample included the entire stream whereas electrofishing estimates were for shorter reaches (typically 50 – 200 m; e.g., Harig and Fausch 1996) usually near the downstream terminus, that are often not representative of all habitat.

*Models of Translocation Success Based on Water Temperature*

Winter water temperatures were not included in higher order models because there was no difference in mean daily temperatures from December through February among cutthroat trout streams with absent, low, or high population status ($P = 0.60$ for ordinal test of independence by PLR). The mean daily water temperature for June through August, which encompasses the period of spawning, egg incubation, and fry emergence for cutthroat trout, ranged from 2.0 to 12.9 °C among study streams in June, 4.2 to 14.6 °C in July (Table 2.4), and 4.9 to 13.9 °C in August. Streams where cutthroat trout populations were classified as absent had significantly colder temperatures than streams with low numbers of cutthroat trout, and both had significantly colder temperatures than streams with high numbers of cutthroat trout for all months and combinations of consecutive months ($P \leq 0.005$ for ordinal test of independence by ordinal PLR) except for June ($P = 0.15$). Examination of the AICc values and their weights ranked mean daily water temperature for July-August combined the highest

(0.289), followed by July (0.256), then August (0.175; Table 2.5). July-August temperatures were only slightly better than July temperatures for predicting translocation success, so July temperatures were chosen to represent summer temperatures in all models. This allowed the largest sample of streams to be used because a thermograph in one stream where temperatures were measured for only one year malfunctioned in August. There was no significant difference in stream temperatures among years ($P = 0.15$ for year effect for 16 streams measured two years or more by two-way ANOVA; Figure 2.2), and the average difference among years was only 0.6 °C (SE = 0.08), so temperatures measured the year after the habitat survey were used in models.

*Models of Translocation Success Based on Water Temperature and Stream-Scale Habitat*

Examination of the AICc values and their weights for 20 nested candidate models based on July mean water temperature and stream-scale habitat data collected during field surveys indicated that the model including summer temperature, bankfull pool width, and number of deep pools was the "best" model for predicting success of cutthroat trout translocations ($P = 0.001$ for ordinal test of independence by PLR; Tables 2.5 and 2.6):

$$P(absent) = \frac{\exp(11.454 - 0.891t - 1.451w - 0.017d)}{1 + \exp(11.454 - 0.891t - 1.451w - 0.017d)}$$

$$P(low + absent) = \frac{\exp(14.077 - 0.891t - 1.451w - 0.017d)}{1 + \exp(14.077 - 0.891t - 1.451w - 0.017d)}$$

where $P$ = probability of a stream having a population status of absent, or low and absent combined, $t$ = mean daily water temperature (°C) for July, $w$ = mean bankfull width of pools (m), and $d$ = total number of deep pools (residual depths ≥ 30 cm). Point estimates

for probability of a stream being in the low and high categories are calculated as $P(\text{low}) = P(\text{low + absent}) - P(\text{absent})$ and $P(\text{high}) = 1 - P(\text{low + absent})$ (Figure 2.3). Translocation streams with a high or low abundance of cutthroat trout had significantly warmer July water temperatures, greater bankfull widths, and more deep pools than streams with fewer cutthroat trout (i.e., high > low > absent; Table 2.4).

The AICc weight for this model was relatively low at 0.286 and was only about twice as likely as the next best model (0.162), so the next three models within two AICc points could be considered competing models (Burnham and Anderson 1998). In this case, a weighted average of the responses from all four models can be used to predict cutthroat trout translocation success, which would provide a more precise prediction than using the "best" model alone (Burnham and Anderson 1998; D. Anderson, Colorado State University, personal communication). However, the four models are similar because they all contain summer water temperature and bankfull pool width (Table 2.6). Moreover, the third variables in three of the models are all measures of the number of pools and are highly correlated with one another ($r \geq 0.95$, $P = 0.001$; see Appendix III). In fact, the predicted values for the "best" model and those from a weighted average of the four competing models differ by only 3%, on average (SE = 0.4; range 0-12%). Therefore, to simplify application by managers, we recommend using only the "best" model to predict cutthroat trout translocation success, recognizing that the variance will be greater than estimated because of the closely weighted competing models.

*Models of Translocation Success Based on Basin-Scale Habitat*

Examination of the AICc values and their weights for the 20 nested candidate models based on basin-scale habitat measured from digital data using a GIS indicated that the model including watershed area and basin aspect was the "best" model for predicting success of cutthroat trout translocations (Table 2.5). However, basin aspect has the opposite sign in the model from that expected (Table 2.7). Theoretically, basin aspect influences the amount of sun exposure and therefore stream temperature, so basins that drain along an East-West axis would receive more sun and have warmer water temperatures than North-South basins. Although we predicted that streams with high numbers of cutthroat trout would have an East-West aspect compared to streams with no cutthroat trout, our data show the opposite (Table 2.4). Streams with no cutthroat trout have an average basin aspect of $35^{o}$ from North (SE = 9), whereas streams with low numbers have an aspect of $36^{o}$ (SE = 7), and streams with high numbers have $22^{o}$ (SE = 4). However, we believe that this result is spurious because of the influence of four large basins with high numbers of cutthroat trout that trend North-South (Figure 2.4), so we excluded this variable from further consideration.

Because the "best" model is not biologically meaningful and is weighted relatively low (0.201), it is not better at explaining the data than other closely weighted models. Therefore, the next three models within 2 AICc points (watershed area, watershed area and basin relief, and watershed area and latitude) should be considered competing models, and a weighted average of their responses used to predict cutthroat trout translocation success (Table 2.7; see Appendix IV). Although model averaging would provide a more precise prediction of translocation success than using only one

model, none of the three competing models based on basin-scale habitat data could detect differences between absent and low status categories ($P \geq 0.24$ for test of independence by binary logistic regression). However, they were able to detect differences between translocation streams in the high versus absent or low categories, so they are useful as a coarse filter.

The three competing models all include watershed area as a critical habitat factor, and basin relief was correlated with watershed area ($r = 0.42$, $P = 0.03$), so the model of watershed area alone may prove most useful as such a coarse filter for predicting translocation success ($P = 0.008$ for ordinal test of independence by PLR; Tables 2.5 and 2.7):

$$P(absent) = \frac{\exp(0.251 - 0.123a)}{1 + \exp(0.251 - 0.123a)}$$

$$P(low + absent) = \frac{\exp(1.804 - 0.123a)}{1 + \exp(1.804 - 0.123a)}$$

where $P$ = probability of a stream having a cutthroat trout population status of absent, or low and absent combined, and $a$ = planimetric watershed area (km$^2$). Although this model cannot detect differences between absent and low population status, a graph of the point estimates can be used to identify a minimum watershed area that has a high probability of supporting high numbers of cutthroat trout (Figure 2.5). For example, translocations into watersheds greater than 14.7 km$^2$ have greater than a 50% chance of establishing high numbers of cutthroat trout.

*Models of Translocation Success Based on Stream- and Basin-Scale Habitat*

Examination of the AICc values and their weights from 15 nested candidate models developed from combinations of habitat variables used in the "best" models with stream- and basin-scale habitat (i.e., July water temperature, pool bankfull width, number of deep pools, watershed area, and basin aspect) indicated that the stream-scale habitat models described previously explained the data better than models using basin-scale habitat attributes (Table 2.5). Models that included basin-scale habitat attributes were not within two AICc points of the "best" model, except for the one including basin aspect, so none were considered competing models.

## DISCUSSION

*Stream Habitat Attributes Limiting Cutthroat Trout*

Greenback and Rio Grande cutthroat trout were originally distributed throughout the large river systems within their historical range (Figure 2.1; Behnke 1992). However, the streams available for translocations have primarily been small, isolated, headwater sites that provide only marginal habitat at best. Many of these streams have failed to sustain robust cutthroat trout populations despite repeated stocking of genetically pure fish. Even though we cannot determine a direct cause-and-effect relationship between habitat and translocation success without an experimental approach, our field surveys of stream-scale habitat indicate that low summer temperature and habitat size are critical factors limiting populations of translocated cutthroat trout.

Our data indicate that many of the greenback and Rio Grande cutthroat trout translocation sites have temperatures that are too cold to support a naturally reproducing

population. Cold summer temperatures are known to delay spawning and prolong egg incubation, resulting in low embryo survival or increased time to fry emergence (Hubert et al. 1994; Stonecypher et al. 1994; Hubert and Gern 1995). Fry that survive and hatch late also may be unable to acclimate quickly enough to rapid drops in water temperature or may starve during winter, so survival may depend on their ability to attain a body size large enough to withstand metabolic deficits (Hunt 1969; Cunjak and Power 1987; Shuter and Post 1990). Therefore, in streams that support low numbers of cutthroat trout, cold temperatures ($\leq 7.8\ ^{o}C$ mean daily temperature for July) likely prevent successful reproduction and recruitment during most summers. Conversely, in streams that support high numbers of cutthroat trout, summer water temperatures are probably warm enough (mean = $10.0^{o}C$; SE= 0.6) to allow successful reproduction in most years.

Our stream-scale habitat model also indicates that habitat size and abundance, as measured by mean pool bankfull width and number of deep pools, are factors limiting cutthroat trout populations. Other studies have found similar positive relationships between stream width and trout presence or abundance (Nelson et al. 1992; Clarkson and Wilson 1995; Kruse et al. 1997; Dunham and Rieman 1999). Furthermore, a manipulative experiment that increased the number of deep pools caused an increased abundance of adult trout in six northern Colorado streams (Riley and Fausch 1995; Gowan and Fausch 1996a), and further research showed that the mechanism involved was movement over long stream reaches (Riley et al. 1992; Gowan and Fausch 1996b). Larger streams can support larger populations, which are less vulnerable to environmental and demographic stochasticity (Lande 1993). Large streams are also more likely to provide enough habitat heterogeneity to meet the diverse habitat needs of

salmonids. For example, juveniles may be able to overwinter in relatively shallow pools or runs in low-elevation streams (Bustard and Narver 1975; Griffith and Smith 1993), but adult trout are believed to need large pool habitats to survive the winter (Cunjak and Power 1986; Chisholm et al. 1987). Therefore, translocation streams that were unable to support a persistent cutthroat trout population probably lacked sufficient or appropriate habitat to promote survival of enough individuals to sustain a population.

Three other models of translocation success with similar variables have similar AICc weights, and indicate that total number of pools and pools with physical structure (large woody debris, boulders, undercut banks) also limit cutthroat trout populations. Other investigations reported that measures of total pool habitat and pools with physical structure are associated with trout presence (Nelson et al. 1992; Young 1996), corroborating our models. A weighted average from all four models could be used to predict translocation success with greater precision than the one we chose, but the responses would differ by only 3%, on average. However, it is unlikely that increasing the precision of the probability of translocation success returned by model averaging is worth the extra time, effort, and money needed to measure physical structure in large numbers of pools ($n = 24 - 571$ for our streams).

*Appropriate Spatial Scales for Isolated Trout Populations*

Studying species distributions over multiple spatial scales to identify the appropriate scale for management has become prevalent in the literature on stream ecology (e.g., Allan et al. 1997; Lohr and Fausch 1997; Armstrong et al. 1998; Torgersen et al. 1999; Labbe and Fausch in press), partially due to the emergence of landscape

ecology (Wiens 1995). Investigations of resident stream salmonids have also expanded in spatial scale since the recognition that a substantial portion of fish in these populations are not sedentary but move, often over great distances (Gowan et al. 1994; Fausch and Young 1995; Young 1996). Traditional habitat studies at the microhabitat scale (see Fausch et al. 1988 for review) have not been able to develop general principles to guide management of fishes, so a coarser-scale approach to stream habitat may be warranted (Schlosser and Angermeier 1995).

We researched two scales of habitat for cutthroat trout, stream and basin scale, which roughly correspond to patch (i.e., local habitat characteristics) and landscape scales (e.g., habitat area) in the ecological literature. Stream-scale models of translocation success indicated that summer water temperature, pool width, and deep pools were critical factors limiting cutthroat trout populations, but models of basin-scale habitat were not as effective for distinguishing between successful and unsuccessful translocations. In a review of studies on a wide array of taxa that considered both patch- and landscape-scales for detecting species presence and abundance, Mazerolle and Villard (1999) found that landscape variables were significant predictors in more than half the studies (59%), but patch characteristics were significant in nearly all the studies (93%). Results are similar for salmonids. Coarse-scale geomorphic variables are good predictors of presence or abundance when they are measured at the reach level (approximates patch scale; e.g., Lanka et al. 1987; Fausch 1989; Clarkson and Wilson 1995). For example, channel slope was identified as a predictor of trout occurrence for stream reaches in the central Rocky Mountains (Chisholm and Hubert 1986; Kozel and Hubert 1989; Kruse et

al. 1997), but mean slope of the entire stream was not important in our models of basin-scale habitat for predicting cutthroat trout translocation success.

One basin-scale habitat attribute that was useful as a coarse filter for translocation success was watershed area. Based on model results, watersheds larger than 14.7 $km^2$ are predicted to have greater than 50% probability of supporting high numbers of cutthroat trout. This result is not surprising considering the wealth of studies, most recently for investigations into metapopulation dynamics (Peltonen and Hanski 1991; Thomas et al. 1992; Wenny et al. 1993; Rieman and McIntyre 1995; Dunham and Rieman 1999), that support the species-area relationship, i.e. the probability that a species will be present in a habitat patch increases with increasing area (MacArthur and Wilson 1967; Diamond 1975). We presume that large watersheds encompass lower elevation habitat that also provides warmer summer water temperatures for cutthroat trout, and have relatively wide stream channels of sufficient length to provide an adequate number of deep pools. It is also probable that large watersheds have sufficient input of large woody debris and boulders to provide physical structure in pools. Only 1 of 6 streams where translocations failed have watershed areas > 14.7 $km^2$, but 8 of 13 streams with high populations have basins ≤ 14.7 $km^2$, so estimating the probability of translocation success in watersheds ≤ 14.7 $km^2$ requires measuring stream-scale habitat throughout the basin in the field. For example, five of the eight latter streams have lakes or large beaver ponds, which likely increase the probability of supporting high numbers of cutthroat trout. Lakes are detectable at the basin-scale, but beaver ponds are ephemeral and not usually marked on maps. Therefore, attributes measured strictly from maps would not be useful for predicting persistence of cutthroat trout populations in small watersheds.

Landscape-scale habitat variables other than habitat area have been useful for some species with large spatial habitat requirements such as mammals with large home ranges (Mladenoff et al. 1999) and birds with high dispersal capabilities (Ganey et al. 1990; Bellamy et al. 1998), but the landscape scale must be properly defined with respect to the taxon under investigation (Mazerolle and Villard 1999). The appropriate scale for predicting the attributes limiting cutthroat trout translocation success is determined by the ecological process being considered (Schlosser and Angermeier 1995), in particular the life history of the cutthroat trout. Greenback and Rio Grande cutthroat trout historically existed in large connected watersheds, which had high habitat heterogeneity that met the diverse habitat required by different life stages and were large enough to support their movements (Young 1995a). However, most populations of these cutthroat trout subspecies are now isolated in small watersheds (5 – 78 $km^2$) and cannot move to a new area if their habitat needs are not met. Isolating cutthroat trout in small basins has removed this mobile component of their life history and reduced their spatial habitat use. Therefore, basin-scale analyses may be too coarse to identify whether an isolated habitat can support cutthroat trout, particularly if many critical habitat attributes are locally controlled. For example, water temperatures in small, mountain streams are difficult to predict without site-specific data (Smith and Lavis 1975). Models of stream temperature are usually complex combinations of stream aspect, riparian vegetation, air temperature, fluctuations in stream discharge, and meteorological data (e.g., Moore 1967; Morse 1970; Smith and Lavis 1975; Bartholow 1993). We found weak correlations between basin-scale measures of elevation or latitude and July water temperatures measured in the field

($r$ = -0.42 and –0.41, respectively, $P \leq 0.03$; Figure 2.6), but neither variable significantly improved upon the model using watershed area alone. Basin aspect was important in the basin-scale models, but the relationship with stream water temperatures was opposite that expected ($r$ = -0.25, $P$ = 0.21; Figure 2.6), supporting the view that its inclusion was a spurious statistical result. Therefore, measuring site-specific variables that would be accurate surrogates for stream temperature would require much more effort and money than simply measuring temperature directly using thermographs.

*Probability of Long-term Persistence*

The short-term (5-25 years) success of a translocation, based on the minimum abundance of cutthroat trout during our survey period, does not necessarily ensure that this population will achieve long-term population persistence (e.g., > 100 years; Rieman and McIntyre 1993). It is assumed that populations established through translocation will be long lasting, but this may not be true for streams supporting low numbers of cutthroat trout. Small populations are at greater risk from demographic and environmental stochasticity (Propst et al. 1992; Lande 1993; Reiman and McIntyre 1993), so it is likely that in many streams with low numbers of trout the population will eventually be extirpated. Furthermore, basin-scale models of cutthroat trout translocation success were unable to detect differences in streams with no trout versus those with low numbers. This suggests that these streams are geomorphically more similar to one another than to streams with high numbers of trout. As such, we suspect that numbers of cutthroat trout in streams with low status may be more likely to dwindle rather than increasing towards the high status category.

*Management Implications*

This research identifies critical habitat attributes that allow successful translocation and at least short-term persistence of cutthroat trout populations in isolated headwater reaches. Managers and biologists can use these habitat models to evaluate potential translocation sites and identify current populations at greatest risk from extirpation. For example, stream habitat was measured in Powderhouse Creek, a site where the translocation was too recent (1997) to assess success. The watershed is only 10 $km^2$, so our basin-scale model is not useful as a coarse filter. However, the stream-scale model for Powderhouse Creek, which has a mean daily July water temperature of 10.0 °C, mean pool bankfull width of 2.2 m, and 5 deep pools, predicts a 33% probability of supporting no trout, a 54% probability of supporting low numbers, and only a 13% chance of high numbers of cutthroat trout (Figure 2.3). Given this, managers will need to decide whether establishing a small population of cutthroat trout is worth the time, money, and effort of a translocation project, considering that it may not achieve long-term persistence.

In contrast, the basin-scale model of translocation success may be useful for identifying current populations at the greatest risk of extirpation. The habitat attributes necessary to establish a population through translocation may not be identical to those that have sustained these historical populations, but managers have limited time and budgets to devote to stream surveys. A coarse filter such as minimum watershed area could be used to prioritize streams. At last count, there were 50 streams with historical, remnant populations of Rio Grande cutthroat trout in New Mexico that remained free from nonnative salmonids (NMDGF unpublished data). Almost half (24) of these 50

populations are in watersheds ≤ 14.7 $km^2$ in area and may warrant first attention (Figure 2.7). Populations in many streams have not been surveyed in 10 or more years and may be at risk of extirpation, particularly if invading nonnative salmonids are reducing their available habitat. Similarly, 4 of 13 historical populations of Rio Grande cutthroat trout and 4 of 7 historical populations of greenback cutthroat trout in Colorado also persist in small watersheds < 14.7 $km^2$ and may be at risk of extirpation from insufficient habitat. None of these smaller watersheds include lakes, which may reduce the probability of extirpation, so field surveys of stream-scale habitat could identify if they have appropriate summer water temperatures, and are wide enough and have a sufficient number of deep pools to support a high abundance of cutthroat trout.

Despite the low success rates for translocations of fishes (e.g., Simons et al. 1989, Harig et al. in review), our research is one of the few attempts to determine the specific factors influencing translocation success (cf. Williams et al. 1988). It demonstrates that measuring attributes of local habitat over a whole watershed scale that matches the life history of the organism can be highly useful for identifying critical habitat factors (Torgersen et al. 1999; Labbe and Fausch in press). The models we developed from stream- and basin-scale habitat attributes will be valuable tools for fisheries managers concerned with the conservation of Rio Grande and greenback cutthroat trout, particularly if included in an active management program that tests and refines these models with data from recent and future translocation sites. Moreover, results from a multi-scale analysis of this type may also be applicable to other subspecies of cutthroat trout in central and southern Rocky Mountain streams (e.g., Colorado River cutthroat trout, *O. c. pleuriticus*) because similar habitat attributes probably limit their populations.

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REFERENCES


Agresti, A. 1996. An introduction to categorical data analysis. John Wiley and Sons, New York, New York, USA.

Allan, J.D., D.L. Erickson, and J. Fay. 1997. The influences of catchment land use on stream integrity across multiple spatial scales. Freshwater Biology 37:149-161.

Alves, J. 1998. Status of Rio Grande cutthroat trout in Colorado. Management Report, Colorado Division of Wildlife, Denver, CO.

Angermeier, P.L., and J.R. Karr. 1984. Relationship between woody debris and fish habitat in a small warmwater stream. Transactions of the American Fisheries Society 113:716-726.

Armstrong, J.D., J.W.A. Grant, H.L. Forsgren, K.D. Fausch, R.M. DeGraaf, I.A. Fleming, T.D. Prowse, and I.J. Schlosser. 1998. The application of science to the management of Atlantic salmon (*Salmo salar*): integration across scales. Canadian Journal of Fisheries and Aquatic Sciences 55(Supplement 1):303-311.

Bartholow, J.M. 1993. Sensitivity of the U.S. Fish and Wildife Service's Stream Network Temperature Model. Pages 247-257 *in* H.J. Morel-Seytoux, H.J., editor). Proceedings of the Thirteenth Annual American Geophysical Union, Fort Collins, Colorado, USA.

Beard, T.D., and R.F. Carline. 1991. Influence of spawning and other stream habitat features on spatial variability of wild brown trout. Transactions of the American Fisheries Society 120:711-722.

Behnke, R.J. 1992. Native trout of western North America. American Fisheries Society Monograph 6, American Fisheries Society, Bethesda, Maryland, USA.

Bellamy, P.E., N.J. Brown, B. Enoksson, L.G. Firbank, R.J. Fuller, S.A. Hinsley, and A.G.M. Schotman. 1998. The influences of habitat, landscape structure and climate on local distribution patterns of the nuthatch (*Sitta europaea* L.). Oecologia 115:127-136.

Benke, A.C., R.L. Henry, D.M. Gillespie, and R.J. Hunter. 1985. Importance of snag habitat for animal production in southeastern streams. Fisheries 10(5):8-13.

Bisson, P.A., J.L. Nielsen, R.A. Palmason, and L.E. Grove. 1981. A system of naming habitat types in small streams, with examples of habitat utilization by salmonids during low streamflow. Pages 62-73 *in* N.B. Armantrout, editor. Acquisition and utilization of aquatic habitat inventory information. American Fisheries Society, Bethesda, Maryland, USA.

Burham, K.P., and D.R. Anderson. 1998. Model selection and inference: a practical information-theoretic approach. Springer-Verlag, New York, New York, USA.

Bustard, D.R., and D.W. Narver. 1975. Preferences of juvenile coho salmon (*Oncorhynchus kisutch*) and cutthroat trout (*Salmo clarki*) relative to simulated winter habitat. Journal of the Fisheries Research Board of Canada 32:681-687.

Chisholm, I.M., and W.A. Hubert. 1986. Influence of stream gradient on standing stock of brook trout in the Snowy Range, Wyoming. Northwest Science 60:137-139.

Chisholm, I.M., W.A. Hubert, and T.A. Wesche. 1987. Winter stream conditions and use of habitat by brook trout in high-elevation Wyoming streams. Transactions of the American Fisheries Society 116:176-184.

Clarkson, R.W., and J.R. Wilson. 1995. Trout biomass and stream habitat relationships in the White Mountains area, east-central Arizona. Transactions of the American Fisheries Society 124:599-612.

Cunjak, R.A., and G. Power. 1986. Winter habitat utilization by resident brook trout (*Salvelinus fontinalis*) and brown trout (*Salmo trutta*). Canadian Journal of Fisheries and Aquatic Sciences 43:1970-1981.

Cunjak, R.A., and G. Power. 1987. The feeding and energetics of stream-resident trout in winter. Journal of Fish Biology 31:493-511.

Diamond, J.M. 1975. Assembly of species and communities. Pages 342-444 *in* M.L. Cody and J.M. Diamond, editors. Ecology and evolution of communities. Harvard University Press, Cambridge, Massachusetts, USA.

Dunham, J.B., and B.E. Rieman. 1999. Metapopulation structure of bull trout: influences of physical, biotic, and geometrical landscape characteristics. Ecological Applications 9:642-655.

Dunham, J.B., and G.L. Vinyard. 1997. Incorporating stream level variability into analyses of site level fish habitat relationships: some cautionary examples. Transactions of the American Fisheries Society 126:232-329.

Dunne, T., and L.B. Leopold. 1978. Water in environmental planning. W.H. Freeman and Company, San Francisco, California, USA.

Eash, D.A. 1994. A Geographic Information System procedure to quantify drainage-basin characteristics. Water Resources Bulletin 30:1-8.

Enderson, J.H., W. Heinrich, L. Kiff, and C.M. White. 1995. Population changes in North American Peregrines. Transactions of the North American Wildlife and Natural Resources Conference 60:142-161.

ESRI (Environmental Systems Research Institute). 1995. ARC/INFO. Version 7.0.3. Redlands, California, USA.

Fausch, K.D. 1984. Profitable stream positions for salmonids: relating specific growth rate to net energy gain. Canadian Journal of Zoology 62:441-451.

Fausch, K.D. 1989. Do gradient and temperature affect distributions of, and interactions between, brook charr (*Salvelinus fontinalis*) and other resident salmonids in streams? Physiological Ecology Japan, Special Volume 1:303-322.

Fausch, K.D. 1993. Experimental analysis of microhabitat selection by juvenile steelhead (*Oncorhynchus mykiss*) and coho salmon (*O. kisutch*) in a British Columbia stream. Canadian Journal of Fisheries and Aquatic Sciences 50:1198-1207.

Fausch, K.D., C. Gowan, A.D. Richmond, and S.C. Riley. 1995. The role of dispersal in trout population response to habitat formed by large woody debris in Colorado mountain streams. Bulletin Français de la Pêche et de la Pisciculture 337/338/339:179-190.

Fausch, K.D., C.L. Hawkes, and M.G. Parsons. 1988. Models that predict the standing crop of stream fish from habitat variables: 1950-85. General Technical Report PNW-213, USDA Forest Service, Portland, Oregon, USA.

Fausch, K.D, and T.G. Northcote. 1992. Large woody debris and salmonid habitat in a small coastal British Columbia stream. Canadian Journal of Fisheries and Aquatic Sciences 49:682-693.

Fausch, K.D., and M.K. Young. 1995. Evolutionary significant units and movement of resident stream fishes: a cautionary tale. American Fisheries Society Symposium 17:360-370.

Franklin, A.B., D.R. Anderson, R.J. Gutierrez, and K.P. Burnham. *In press*. Climate, habitat quality, and fitness in northern spotted owl populations in northwestern California. Ecological Monographs.

Flebbe, P.A., and C.A. Dolloff. 1995. Trout use of woody debris and habitat in Appalachian wilderness streams of North Carolina. North American Journal of Fisheries Management 15:579-590.

Frissell, C.A., W.J. Liss, C.E. Warren, and M.D. Hurley. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. Environmental Management 10:199-214.

Ganey, J.L., M.E. Alderson, R.P. Balda, and D.R. Patton. 1990. Developing a large-scale model for Mexican spotted owl habitat. Pages 182-190 *in* P.R. Krausman

and N.S. Smith, editors. Managing wildlife in the southwest. Arizona Chapter of the Wildlife Society, Phoenix, Arizona, USA.

Gordon, D.R. 1994. Translocation of species into conservation areas: a key for natural resource managers. Natural Areas Journal 14:13-37.

Gowan, C., and K.D. Fausch. 1996a. Long-term demographic responses of trout populations to habitat manipulation in six Colorado streams. Ecological Applications 6:931-946.

Gowan, C., and K.D. Fausch. 1996b. Mobile brook trout in two high elevation Colorado streams: re-evaluating the concept of restricted movement. Canadian Journal of Fisheries and Aquatic Sciences 53:1370-1381.

Gowan, C., M.K. Young, K.D. Fausch, and S.C. Riley. 1994. Restricted movement in resident stream salmonids: a paradigm lost? Canadian Journal of Fisheries and Aquatic Sciences 51:2626-2637.

Gresswell, R.E. 1988. Status and management of interior stocks of cutthroat trout. American Fisheries Society, Symposium 4, Bethesda, Maryland, USA.

Griffith, B., J.M. Scott, J.W. Carpenter, and C. Reed. 1989. Translocation as a species conservation tool: status and strategy. Science 245:477-480.

Griffith, J.S. 1972. Comparative behavior and habitat utilization of brook trout (*Salvelinus fontinalis*) and cutthroat trout (*Salmo clarki*) in small streams in northern Idaho. Journal of Fisheries Research Board of Canada 29:265-273.

Griffith, J.S., and R.W. Smith. 1993. Use of winter concealment cover by juvenile cutthroat trout and brown trout in the South Fork of the Snake River, Idaho. North American Journal of Fisheries Management 13:823-830.

Harig, A.L., and K.D. Fausch. 1996. Compilation of data on Colorado waters containing Rio Grande cutthroat trout. Technical Report No. 904-96, Colorado Division of Wildlife, Montrose, Colorado, USA.

Harig, A.L., K.D. Fausch, and M.K. Young. *In review*. Factors influencing success of translocated greenback cutthroat trout populations. North American Journal of Fisheries Management.

Hawkins, C.P., J.L. Kershner, P.A. Bisson, M.D. Bryant, L.M. Decker, S.V. Gregory, D.A. McCullough, C.K. Overton, G.H. Reeves, R.J. Steedman, and M.K. Young. 1993. A hierarchical approach to classifying stream habitat features. Fisheries 18(6):3-11.

Heggenes, J., and T. Traaen. 1988. Downstream migration and critical water velocities in stream channels for fry of four salmonid species. Journal of Fish Biology 32:717-727.

Hendrickson, D.A. and J.E. Brooks. 1991. Transplanting short-lived fishes in North American deserts: review, assessment, and recommendations. Pages 283-298 *in* W.L. Minckley and J.E. Deacon, editors. Battle against extinction: native fish management in the American West. University of Arizona Press, Tucson, Arizona, USA.

Hicks, B.J., J.D. Hall, P.A. Bisson, and J.R. Sedell. 1991. Responses of salmonids to habitat changes. Pages 483-518 *in* W.R. Meehan, editor. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society Special Publication 19, Bethesda, Maryland, USA.

Hodder, K.H., and J.M. Bullock. 1997. Translocations of native species in the UK: implications for biodiversity. Journal of Applied Ecology 34:547-565.

Hubert, W.A., and W.A. Gern. 1995. Influence of embryonic stage on survival of cutthroat trout exposed to temperature reduction. The Progressive Fish-Culturist 57:326-328.

Hubert, W.A., R.W. Stonecypher, W.A. Gern, and J. Bobbit. 1994. Response of cutthroat trout embryos to reduced incubation temperatures at different developmental stages. The Progressive Fish-Culturist 56:185-187.

Hughes, N.F., and L.M. Dill. 1990. Position choice by drift-feeding salmonids: model and test for Arctic grayling (*Thymallus arcticus*) in subarctic mountain streams, interior Alaska. Canadian Journal of Fisheries and Aquatic Sciences 47:2039-2048.

Hunt, R.L. 1969. Overwinter survival of wild fingerling brook trout in Lawrence Creek, Wisconsin. Journal of Fisheries Research Board of Canada 26:1473-1483.

IUCN (International Union for Conservation of Nature and Natural Resources). 1987. Position statement on translocations of living organisms: introductions, reintroductions, and re-stocking. IUCN Council, Gland, Switzerland.

IUCN (International Union for Conservation of Nature and Natural Resources). 1995. Guidelines for re-introductions. IUCN Council, Gland, Switzerland.

Kleiman, D.G. 1989. Reintroduction of captive mammals for conservation: guidelines for reintroducing endangered species into the wild. BioScience 39:152-161.

Kozel, S.J., and W.A. Hubert. 1989. Factors influencing the abundance of brook trout (*Salvelinus fontinalis*) in forested mountain streams. Journal of Freshwater Ecology 5:113-122.

Kruse, C.G., W.A. Hubert, and F.J. Rahel. 1997. Geomorphic influences on the distribution of Yellowstone cutthroat trout in the Absaroka Mountains, Wyoming. Transactions of the American Fisheries Society 126:418-427.

Labbe, T. R., and K. D. Fausch. *In press*. Dynamics of intermittent stream habitat regulate persistence of a threatened fish at multiple scales. Ecological Applications.

Lande, R. 1993. Risks of population extinction from demographic and environmental stochasticity and random catastrophes. The American Naturalist 142:911-927.

Lanka, R.P., W.A. Hubert, and T.A. Wesche. 1987. Relations of geomorphology to stream habitat and trout standing stock in small Rocky Mountain streams. Transactions of the American Fisheries Society 116:21-28.

Latterell, J. J., K.D. Fausch, C. Gowan, and S.C. Riley. 1998. Relationship of trout recruitment to snowmelt runoff flows and adult trout abundance in six Colorado mountain streams. Rivers 6:240-250.

Lisle, T.E. 1987. Using "residual depths" to monitor pool depths independently of discharge. Research Note No. PSW-394, USDA Forest Service, Berkeley, California, USA.

Lohr, S.C., and K.D. Fausch. 1997. Multiscale analysis of natural variability in stream fish assemblages of a western Great Plains watershed. Copeia 4:706-724.

MacArthur, R.H., and E.O. Wilson. 1967. The theory of island biogeography. Princeton University Press, Princeton, New Jersey, USA.

Mark, D.M. 1975. Geomorphic parameters: a review and evaluation. Geografiska Annaler 57A:165-177.

Mazerolle, M.J., and M. Villard. 1999. Patch characteristics and landscape context as predictors of species presence and abundance: a review. Ecoscience 6:117-124.

Millsap, B.A., P.L. Kennedy, M.A. Byrd, G. Court, J.H. Enderson, and R.N. Rosenfield. 1998. Scientific review of the proposal to de-list the American peregrine falcon. Wildlife Society Bulletin 26:522-538.

Minckley, W.L. 1995. Translocation as a tool for conserving imperiled fishes: experiences in western United States. Biological Conservation 72:297-309.

Mladenoff, D.J., T.A. Sickley, and A.P. Wydeven. 1999. Predicting gray wolf landscape recolonization: logistic regression models versus new field data. Ecological Applications 9:37-44.

Moore, A.M. 1967. Correlation and analysis of water-temperature data for Oregon streams. U.S. Geological Survey Water-Supply Paper 1819-K, Washington, D.C., USA.

Moore, I.D., R.B. Grayson, and A.R. Ladson. 1991. Digital Terrain Modelling: a review of hydrological, geomorphological, and biological applications. Hydrological Processes 5:3-30.

Morse, W.L. 1970. Stream temperature prediction model. Water Resources Research 6:290-302.

Moyle, P.B., and G.M. Sato. 1991. On the design of preserves to protect native fishes. Pages 155-170 *in* W.L. Minckley and J.E. Deacon, editors. Battle against extinction: native fish management in the American West. University of Arizona Press, Tucson, Arizona, USA.

Moyle, P.B., and R.M. Yoshiyama. 1994. Protection of aquatic biodiversity in California: a five-tiered approach. Fisheries 19(2):6-18.

Nakano, S., S. Kitano, K. Nakai, and K.D. Fausch. 1998. Competitive interactions for foraging microhabitat among introduced brook charr, *Salvelinus fontinalis*, and native bull charr, *S.confluentis*, and westslope cutthroat trout, *Oncorhynchus clarki lewisi*, in a Montana stream. Environmental Biology of Fishes 52:345-355.

Nehring, R.B., and R.M. Anderson. 1993. Determination of population-limiting critical salmonid habitat in Colorado streams using the Physical Habitat Simulation System. Rivers 4:1-19.

Nelson, R.L., W.S. Platts, D.P. Larsen, and S.E. Jensen. 1992. Trout distribution and habitat in relation to geology and geomorphology in the North Fork Humboldt River drainage, northeastern Nevada. Transactions of the American Fisheries Society 121:405-426.

ODFW (Oregon Department of Fish and Wildlife). 1994. Methods for stream habitat surveys. Aquatic Inventory Project Research Section, Oregon Department of Fish and Wildlife, Corvallis, Oregon, USA.

Otis, D.L., K.P. Burnham, G.C. White, and D.R. Anderson. 1978. Statistical inference from capture data on closed animal populations. Wildlife Monographs 62.

Overton, C.K., J.D. McIntyre, R. Armstrong, S.L. Whitwell, and K.A. Duncan. 1995. User's guide to fish habitat: descriptions that represent natural conditions in the

Salmon River basin, Idaho. General Technical Report INT-GTR-322, USDA Forest Service, Ogden, Utah, USA.

Peltonen, A., and I. Hanski. 1991. Patterns of island occupancy explained by colonization and extinction rates in shrews. Ecology 72:1698-1708.

Platts, W.S. 1979. Relationships among stream order, fish populations, and aquatic geomorpholgy in an Idaho river drainage. Fisheries 4(2):5-9.

Poff, N.L. 1997. Landscape filters and species traits: towards mechanistic understanding and prediction in stream ecology. Journal of the North American Benthological Society 16:391-409.

Poff, N.L., and J.V. Ward. 1989. Implications of streamflow variability and predictability for lotic community structure: a regional analysis of streamflow patterns. Canadian Journal of Fisheries and Aquatic Sciences 46:1805-1818.

Rieman, B.E., and J.D. McIntyre. 1993. Demographic and habitat requirements for conservation of bull trout. General Technical Report INT-302, USDA Forest Service, Ogden, Utah, USA.

Rieman, B.E., and J.D. McIntyre. 1995. Occurrence of bull trout in naturally fragmented habitat patches of varied size. Transactions of the American Fisheries Society 124:285-296.

Richmond, A.D., and K.D. Fausch. 1995. Characteristics and function of large woody debris in subalpine Rocky Mountain streams in northern Colorado. Canadian Journal of Fisheries and Aquatic Sciences 52:1789-1802

Riley, S.C., and K.D. Fausch. 1995. Trout population response to habitat manipulation in six northern Colorado streams. Canadian Journal of Fisheries and Aquatic Sciences 52:34-53.

Riley, S.C., K.D. Fausch, and C. Gowan. 1992. Movement of brook trout (*Salvelinus fontinalis*) in four small subalpine streams in northern Colorado. Ecology of Freshwater Fish 1:112-122.

Rinne, J.N. 1980. Spawning habitat and behavior of Gila trout, a rare salmonid of the Southwestern United States. Transactions of the American Fisheries Society 109:83-91.

Ruth, T.K., K.A. Logan, L.L. Sweanor, M.G. Hornhocker, and L.J. Templeton. 1998. Evaluating cougar translocation in New Mexico. Journal of Wildlife Management 62:1264-1275.

SAS Institute Inc. 1996. SAS Procedures Guide for Personal Computers, Version 6.12. Cary, NC.

Schlosser, I.J. 1991. Stream fish ecology: a landscape perspective. BioScience 41:704-712.

Schlosser, I.J. 1995. Critical landscape attributes that influence fish population dynamics in headwater streams. Hydrobiologia 303:71-81.

Schlosser, I.J., and P.L. Angermeier. 1995. Spatial variation in demographic processes of lotic fishes: conceptual models, empirical evidence, and implications for conservation. American Fisheries Society Symposium 17:392-401.

Scrivener, J.C., and M.J. Brownlee. 1989. Effects of forest harvesting on spawning gravel and incubation survival of chum (*Oncoryhnchus keta*) and coho salmon (*O. kisutch*) in Carnation Creek, British Columbia. Canadian Journal of Fisheries and Aquatic Sciences 46:681-696.

Shuler, S.W., R.B. Nehring, and K.D. Fausch. 1994. Diel habitat selection by brown trout in the Rio Grande River, Colorado, after placement of boulder structures. North American Journal of Fisheries Management 14:99-111.

Shuter, B.J., and J.R. Post. 1990. Climate, population viability, and the zoogeography of temperate fishes. Transactions of the American Fisheries Society 119:314-336.

Simons, L. H., D. A. Hendrickson, and D. Papoulias. 1989. Recovery of the Gila topminnow: a success story? Conservation Biology 3:11-15.

Smith, K., and M.E. Lavis. 1975. Environmental influences on the temperature of a small upland stream. Oikos 26:228-236.

Stonecypher, R.W., W.A. Hubert, and W.A. Gern. 1994. Effect of reduced incubation temperatures on survival of trout embryos. The Progressive Fish-Culturist 56:180-184.

Struhsaker, T.T., and K.S. Siex. 1998. Translocation and introduction of the Zanzibar red colobus monkey: success and failure with an endangered island endemic. Oryx 32:277-284.

Stuber, R.J., B.D. Rosenlund, and J.R. Bennett. 1988. Greenback cutthroat trout recovery program: management overview. American Fisheries Society Symposium 4:71-74.

Stumpff, W.K., and J. Cooper. 1996. Rio Grande cutthroat trout (*Oncorhynchus clarki virginalis*). Pages 74-86 *in* D.A. Duff, editor. Conservation assessment for inland

cutthroat trout: distribution, status, and habitat management implications. USDA Forest Service, Ogden, Utah, USA.

Tear, T.H., J.M. Scott, P.H. Hayward, and B. Griffith. 1993. Status and prospects for success of the Endangered Species Act: a look at recovery plans. Science 262:976-977.

Thomas, C.D., J.A. Thomas, and M.S. Warren. 1992. Distributions of occupied and vacant butterfly habitats in fragmented landscapes. Oecologia 92:563-567.

Thurow, R.T., and J.G. King. Attributes of Yellowstone cutthroat trout redds in a tributary of the Snake River, Idaho. Transactions of the American Fisheries Society 123:37-50.

Torgersen, C. E., D. M. Price, H. W. Li, and B. A. McIntosh. 1999. Multiscale thermal refugia and stream habitat associations of chinook salmon in northeastern Oregon. Ecological Applications 9:301-319.

USFWS (U.S. Fish and Wildlife Service). 1998. Greenback cutthroat trout recovery plan. U.S. Fish and Wildlife Service, Denver, Colorado, USA.

Wang, L., and R.J. White. 1994. Competition between wild brown trout and hatchery greenback cutthroat trout of largely wild parentage. North American Journal of Fisheries Management 14:475-487.

Weaver, T.M., and J.J. Fraley. 1993. A method to measure emergence success of Westslope cutthroat trout fry from varying substrate compositions in a natural stream channel. North American Journal of Fisheries Management 13:817-822.

Wenny, D.G., R.L. Clawson, J. Faaborg, and S.L. Sheriff. 1993. Population density, habitat selection and minimum area requirements for three forest-interior warblers in central Missouri. Condor 95:968-979.

Wiens, J.A. 1995. Landscape mosaics and ecological theory. Pages 1-26 *in* L. Hansson, L. Fahrig, and G. Merriam, editors. Mosaic landscapes and ecological processes. Chapman and Hall, London, UK.

Williams, J. E., and 12 coauthors. 1988. American Fisheries Society guidelines for introductions of threatened and endangered fishes. Fisheries 13(5):5-11.

Wilzbach, M.A. 1985. Relative roles of food abundance and cover in determining the habitat distribution of stream-dwelling cutthroat trout (*Salmo clarki*). Canadian Journal of Fisheries and Aquatic Sciences 42:1668-1672.

Wolf, C.M., T. Garland, and B. Griffith. 1998. Predictors of avian and mammalian translocation success: reanalysis with phylogenetically independent contrasts. Biological Conservation 86:243-255.

Wolf, C.M., B. Griffith, C. Reed, and S.A. Temple. 1996. Avian and mammalian translocations: update and reanalysis of 1987 survey data. Conservation Biology 10:1142-1154.

Young, M.K., editor. 1995a. Conservation assessment for inland cutthroat trout. General Technical Report RM-256, USDA Forest Service, Fort Collins, Colorado, USA.

Young, M.K. 1995b. Synthesis of management and research considerations. Pages 55-61 *in* M.K. Young, editor. Conservation assessment for inland cutthroat trout. General Technical Report RM-256, USDA Forest Service, Fort Collins, Colorado, USA.

Young, M.K. 1996. Summer movements and habitat use by Colorado River cutthroat trout (*Oncoryhnchus clarki pleuriticus*) in small, montane streams. Canadian Journal of Fisheries and Aquatic Sciences 53:1403-1408.

Young, M.K., R.N. Schmal, T.W. Kohley, and V.G. Leonard. 1996. Conservation status of Colorado River cutthroat trout. General Technical Report RM-GTR-282, USDA Forest Service, Fort Collins, Colorado, USA.

Table 2.1. Characteristics of the 28 study streams where cutthroat trout were translocated, including subspecies (GB = greenback cutthroat trout, RG = Rio Grande cutthroat trout), location (NF = National Forest, RMNP = Rocky Mountain National Park), latitude (degrees N), longitude, elevation of the lower stream terminus, stream length, mean stream gradient, and potential barrier to fish movement. Latitude, longitude, minimum elevation, and stream gradient were identified using a GIS. Stream length was the amount of stream habitat available in summer, calculated from a USGS 1:24,000-scale quadrangle as the distance from the lower fish movement barrier to the upper terminus of the field survey. Potential barriers are described as height above the water surface (h), reach gradient, or reach length (l). Heights of waterfalls and artificial structures were measured in the field (estimated for waterfalls > 7 m), whereas reach gradient and length were measured from a USGS 1:24,000-scale quadrangle.

| No | Stream | Sub-species | State | Location | Latitude | Longitude | Minimum elevation (m) | Stream length (km) | Mean gradient (%) | Potential barrier to fish movement |
|---|---|---|---|---|---|---|---|---|---|---|
| 1 | Benito Creek | RG | CO | Rio Grande NF | 37° 56′ 30″ | 106° 48′ 08″ | 3281 | 2.6 | 13.3 | Steep gradient (~ 36%) |
| 2 | Cony Creek | GB | CO | RMNP | 40° 11′ 30″ | 105° 35′ 27″ | 2904 | 4.9[a] | 10.5 | Steep cascades (gradient ~ 25%) |
| 3 | Cottonwood Creek | GB | CO | Pike-San Isabel NF | 38° 06′ 00″ | 105° 31′ 14″ | 2470 | 5.3 | 10.8 | Dry stream channel (l > 1.0 km) |
| 4 | Doctor Creek | RG | NM | Santa Fe NF | 35° 46′ 07″ | 105° 42′ 19″ | 2520 | 3.5 | 20.2 | Waterfall (h = 1.4 m) |
| 5 | East Middle Creek | RG | CO | Rio Grande NF | 38° 19′ 07″ | 106° 16′ 45″ | 2939 | 3.5 | 14.8 | Steep gradient (~22%) |

Table 2.1. Continued.

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| 6 | Fern Creek | GB | CO | RMNP | 40° 20′ 38″ | 105° 40′ 13″ | 2818 | 2.4[a] | 17.0 | Waterfall (h ~ 10 m) |
| 7 | Greenhorn Creek | GB | CO | Pike-San Isabel NF | 37° 55′ 23″ | 105° 01′ 44″ | 3107 | 3.4 | 16.6 | Series of falls (h ≥ 2.1 m) |
| 8 | Hourglass Creek | GB | CO | Arapaho-Roosevelt NF | 40° 34′ 60″ | 105° 38′ 36″ | 2861 | 2.8 | 12.3 | Fish weir (h = 1.0 m) |
| 9 | Jacks Creek | RG | NM | Santa Fe NF | 35° 49′ 39″ | 105° 39′ 22″ | 2531 | 11.0 | 11.5 | Waterfall (h = 1.9 m) |
| 10 | Little Medano Creek | RG | CO | Great Sand Dunes National Monument | 37° 47′ 59″ | 105° 31′ 04″ | 2588 | 3.1 | 16.5 | Dry stream channel (l = 1.7 km) |
| 11 | Little Ute Creek | RG | CO | Private land | 37° 33′ 49″ | 105° 26′ 29″ | 3262 | 2.2 | 16.0 | Series of falls (h ≥ 3.0 m) |
| 12 | May Creek | GB | CO | Arapaho-Roosevelt NF | 40° 36′ 26″ | 105° 47′ 44″ | 2718 | 4.7 | 12.2 | Steep gradient (~22%) |
| 13 | Medano Creek and tributaries | RG | CO | Great Sand Dunes National Monument | 37° 45′ 59″ | 105° 30′ 21″ | 2502 | 20.5 | 13.2 | Dry stream channel (l > 1.0 km) |

Table 2.1. Continued.

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| 14 | Nabor Creek | RG | CO/ NM | NM State Wildlife Area and private land | 36° 58′ 37″ | 106° 37′ 39″ | 2545 | 5.6[a] | 12.7 | Rock gabion (h = 1.4 m) |
| 15 | North Fork Big Thompson River | GB | CO | RMNP | 40° 30′ 11″ | 105° 35′ 60″ | 3293 | 1.8[a] | 13.6 | Waterfall (h = 1.6 m) |
| 16 | Ouzel Creek | GB | CO | RMNP | 40° 11′ 39″ | 105° 38′ 17″ | 3159 | 1.0 | 17.3 | Series of falls (h ≥ 3.0 m) |
| 17 | Pecos River | RG | NM | Santa Fe NF | 35° 56′ 13″ | 105° 33′ 28″ | 3178 | 5.8 | 8.4 | Waterfall (h ~ 9 m) |
| 18 | Powderhouse Creek | RG | NM | Carson NF | 36° 51′ 58″ | 105° 16′ 20″ | 2928 | 4.3 | 10.6 | Wooden dam (h = 1.5 m) |
| 19 | Rio Cebolla | RG | NM | Santa Fe NF | 35° 57′ 16″ | 106° 40′ 17″ | 2495 | 6.7[a] | 8.8 | Earthen dam and culvert (h = 4.0 m) |
| 20 | Rio de los Pinos | RG | CO | Rio Grande NF | 37° 04′ 55″ | 106° 29′ 47″ | 3206 | 3.1 | 12.8 | Waterfall (h ~ 15 m) |
| 21 | Roaring River | GB | CO | RMNP | 40° 25′ 53″ | 105° 38′ 22″ | 2881 | 6.9[a] | 16.4 | Steep cascades (gradient ~ 49%) |

Table 2.1. Concluded.

| | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|
| 22 | Rough Canyon/ Rhodes Gulch | RG | CO | Rio Grande NF | 37° 23′ 15″ | 106° 24′ 35″ | 2738 | 4.9 | 13.4 | Mining pollutants in receiving waters |
| 23 | San Francisco Creek | RG | CO | Private land and Rio Grande NF | 37° 40′ 02″ | 106° 19′ 57″ | 2400 | 12.4 | 9.1 | Water temperatures too warm for cutthroat trout |
| 24 | Sheep Creek | GB | CO | Arapaho-Roosevelt NF | 40° 39′ 37″ | 105° 44′ 45″ | 2828 | 11.6 | 9.5 | Steep cascades (gradient ~ 45%) |
| 25 | Unknown Creek | RG | CO | Rio Grande NF | 37° 55′ 39″ | 106° 42′ 28″ | 3125 | 3.7 | 7.3 | Steep gradient (~ 21%) |
| 26 | West Creek | GB | CO | RMNP | 40° 27′ 06″ | 105° 31′ 01″ | 2496 | 5.0 | 17.0 | Waterfall (h = 6.9 m) |
| 27 | Williams Gulch | GB | CO | Arapaho-Roosevelt NF | 40° 42′ 42″ | 105° 45′ 56″ | 2758 | 4.1 | 7.3 | Steep cascades (gradient ~ 43%) |
| 28 | West Fork San Francisco Creek | RG | CO | Rio Grande NF | 37° 33′ 22″ | 106° 23′ 44″ | 2890 | 4.6 | 14.8 | Rock gabion (h = 0.4 m) |

[a] These stream habitats included one or more lakes that support cutthroat trout.

Table 2.2. Habitat factors included in stream-scale models calculated from variables measured during basin-wide field surveys of translocation streams.

| Habitat Variable | Definition |
|---|---|
| Stream length | Length (km) of stream surveyed from the fish movement barrier upstream to the end of pool habitat (< 2.0 m bankfull width), measured from a 1:24,000 USGS topographic quadrangle. |
| Number of pools | Total number of pools at least ½ channel width long with residual depths ≥ 18 cm. Pools were identified according to Hawkins et al. (1993) as channel geomorphic units formed by interactions among discharge, sediment load, and channel resistance to flow. |
| Bankfull pool width | Grand mean bankfull width (m) of all pools, calculated from measures at the downstream, center, and upstream ends of each pool, at the height where the water surface is level with the floodplain (Dunne and Leopold 1978). |
| Deep pools | Number of pools with residual depth ≥ 30 cm, calculated from the maximum depth minus the maximum tail crest depth measured at the downstream hydraulic control that forms the pool (Lisle 1987). This depth criterion was based on the median residual depth of all pools surveyed in all streams. |

Table 2.2. Concluded.

| | |
|---|---|
| Large woody debris | Number of pools with at least one piece of large woody debris, which was at least 15 cm in diameter for 3 m of length (adapted from Richmond and Fausch 1995) and at least partially in or suspended over the bankfull channel (including pieces forming pools). |
| Boulders | Number of pools with at least one boulder, which was >50 cm in diameter in all dimensions and within the bankfull channel, including those forming the stream bank if they protruded into the pool. |
| Undercut bank | Number of pools with at least 0.2 m of undercut bank, which was at least 10 cm undercut, no more than 15 cm above the water surface, and had a minimum water depth of 10 cm (adapted from Fausch and Northcote 1992). |
| Clean gravel | Number of pools with at least 25% area as clean gravel (6-63 mm in diameter, free from silt) in the downstream quarter of the pool, estimated visually. |

Table 2.3. Basin-scale habitat factors measured for each translocation stream using a Geographic Information System and digital data derived from USGS digital elevation models. Specific commands from the Arc/Info® hydrological modeling tools (ESRI 1995) are listed in capitals letters.

| Habitat Variable | Definition |
|---|---|
| Watershed area | Total upslope area ($km^2$) contributing flow to the basin outlet, calculated as planimetric watershed area (ZONALGEOMETRY). |
| Main channel length | Surface length (km) measured along the main channel (excluding tributaries) from the basin outlet to the headwaters. The GIS defined channels in cells where flow accumulation exceeded 1000 upslope cells, and incorporated changes in elevation in its calculation (SURFACELENGTH). |
| Total channel length | Surface length (km) computed by summing the length of all stream segments within the drainage basin, including tributary streams, which may also provide trout habitat. Channels were defined as for main channel length (SURFACELENGTH). |
| Drainage density | The total length of stream channels per unit basin area ($km/km^2$), calculated as total channel length divided by watershed area. |
| Latitude | The latitude of the basin outlet in UTM coordinates (m; CELLVALUE). |
| Elevation | Minimum elevation above sea level (m), measured at the basin outlet (ZONALSTATS). |

Table 2.3. Concluded.

| | |
|---|---|
| Basin relief | The difference between the highest and lowest elevations occurring within a basin (m; ZONALSTATS). |
| Basin slope | The percentage change in elevation quantified for each basin as mean percent rise (ZONALSTATS). |
| Basin aspect | The mean direction of the drainage basin measured as compass degrees either clockwise or counterclockwise from North. Data were restricted to values between $0^{\circ}$ and $90^{\circ}$, which correspond to a North-South versus East-West aspect, respectively (ZONALSTATS). |
| Stream slope | The percentage change in elevation quantified for each stream network as mean percent rise (ZONALSTATS). |
| Stream aspect | The mean direction of the stream network measured as compass degrees either clockwise or counterclockwise from North. Data were restricted to values between $0^{\circ}$ and $90^{\circ}$, which correspond to a North-South versus East-West aspect, respectively (ZONALSTATS). |

Table 2.4. Values for habitat variables for the 28 study streams where cutthroat trout were translocated. Variables shown are used in the "best" models to predict probability of cutthroat trout translocation success. Total number of pools and number of pools with structure were highly correlated with number of deep pools ($r \geq 0.95$ for both), so are not shown here (see text and Appendix III).

| Stream | Number of age-1 and older trout basin-wide | Mean July daily temperature (°C) | Mean bankfull pool width (m) | Number of deep pools (RD ≥ 30 cm) | Watershed area ($km^2$) | Basin aspect (degrees from N) |
|---|---|---|---|---|---|---|
| *Absent* | | | | | | |
| Benito Creek | 0 | 4.8 | 1.8 | 12 | 5.1 | 19 |
| Doctor Creek | 0 | 10.2 | 3.2 | 22 | 7.1 | 58 |
| Hourglass Creek | 0 | 4.2 | 3.8 | 94 | 9.5 | 64 |
| Little Medano Cr. | 3 | 7.1 | 2.1 | 35 | 16.2 | 10 |
| Unknown Creek | 1 | 6.5 | 1.0 | 2 | 8.5 | 32 |
| W.F. San Francisco Cr. | 0 | 9.5 | 2.8 | 13 | 7.2 | 25 |
| Mean (SE) | - | 7.1 (1.0) | 2.5 (0.4) | 30 (14) | 8.9 (1.6) | 35 (9) |
| *Low Population* | | | | | | |
| Cottonwood Creek | 67 | 7.1 | 2.6 | 131 | 9.2 | 41 |
| Little Ute Creek | 16 | 7.8 | 4.5 | 22 | 8.9 | 48 |
| May Creek | 65 | 8.4 | 3.3 | 144 | 12.5 | 59 |
| Ouzel Creek | 36 | 6.0 | 4.1 | 12 | 7.0 | 15 |
| Pecos River | 26 | 9.2 | 2.9 | 61 | 11.3 | 18 |

Table 2.4. Continued.

| | | | | | | |
|---|---|---|---|---|---|---|
| Rio de los Pinos | 49 | 8.7 | 3.2 | 36 | 9.5 | 26 |
| Rough C./ Rhodes G. | 40 | 8.3 | 2.7 | 29 | 10.3 | 18 |
| West Creek | 82 | 7.1 | 4.1 | 117 | 25.3 | 67 |
| Mean (SE) | - | 7.8 (0.4) | 3.4 (0.3) | 69 (19) | 11.7 (2.0) | 36 (7) |
| *High Population* | | | | | | |
| Cony Creek | 129[a, b] | 9.0 | 5.4 | 146 | 14.5 | 17 |
| East Middle Creek | 117[b] | 9.7 | 3.2 | 46 | 14.2 | 43 |
| Fern Creek | 61[b] | 7.7 | 4.5 | 56 | 6.8 | 17 |
| Greenhorn Creek | 173 | 8.4 | 2.9 | 121 | 7.3 | 17 |
| Jacks Creek | 322[c] | 9.8 | 3.6 | 197 | 18.4 | 4 |
| Medano Creek | 1278 | 10.5 | 4.1 | 361 | 77.8 | 8 |
| Nabor Creek | 719[b] | 14.5 | 3.8 | 74 | 11.9 | 23 |
| N.F. Big Thompson R. | 112[b] | 8.1 | 3.1 | 18 | 8.3 | 47 |
| Rio Cebolla | 778[c] | 14.6 | 3.0 | 69 | 37.1 | 20 |
| Roaring River | 124[a] | 9.9 | 4.9 | 21 | 14.6 | 14 |
| San Francisco Creek | 193[a] | 8.8 | 3.9 | 88 | 42.5 | 14 |
| Sheep Creek | 661 | 7.8 | 4.1 | 318 | 36.1 | 24 |
| Williams Gulch | 319 | 10.8 | 2.3 | 105 | 9.0 | 32 |

Table 2.4. Concluded.

| | | | | | | |
|---|---|---|---|---|---|---|
| Mean (SE) | - | 10.0 (0.6) | 3.8 (0.2) | 125 (30) | 22.9 (5.7) | 22 (4) |
| *Test Translocation Site* | | | | | | |
| Powderhouse Creek | _ | 10.0 | 2.2 | 5 | 10.0 | 27 |

[a] Minimum abundance of age-1 and older trout was probably underestimated due to deep, turbulent, or turbid water.

[b] Minimum abundance of age-1 and older trout was probably underestimated due to a lake or large beaver pond where it was not possible to make accurate visual counts.

[c] Minimum abundance of age-1 and older trout was probably underestimated due to thick riparian or aquatic vegetation.

Table 2.5. Statistical models predicting cutthroat trout translocation success from summer water temperatures measured using thermographs, water temperature and stream-scale habitat collected during field surveys, basin-scale habitat measured from digital data, and a combination of stream- and basin-scale factors from the "best" models. Akaike's Information Criterion corrected for small sample size (AICc) and AIC weights (*w*) were used to select the "best approximating" models from each set of *a priori* candidate models. Each set of models is nested but arranged according to their AICc ranking.

| Model | AICc | *w* |
|---|---|---|
| *Summer Water Temperature* | | |
| July through August | 51.420 | 0.289 |
| July | 51.656 | 0.256 |
| August | 52.416 | 0.175 |
| June through August | 55.365 | 0.040 |
| Intercept only | 61.014 | 0.002 |
| June | 61.476 | 0.002 |
| *Water Temperature and Stream-Scale Habitat* | | |
| Summer temperature, pool width, number of deep pools | 45.518 | 0.286 |
| Summer temperature, pool width | 46.678 | 0.162 |
| Summer temperature, pool width, number of all pools | 46.997 | 0.138 |
| Summer temperature, pool width, number of pools with structure | 47.298 | 0.118 |
| Summer temperature, pool width, stream length | 48.231 | 0.074 |
| Summer temperature, pool width, number of pools with clean gravel | 48.354 | 0.070 |

Table 2.5. Continued.

| | | |
|---|---|---|
| Summer temperature, number of deep pools | 48.726 | 0.058 |
| Summer temperature, number of all pools | 50.904 | 0.020 |
| Pool width, number of all pools | 51.376 | 0.015 |
| Summer temperature | 51.656 | 0.013 |
| Pool width, stream length | 51.791 | 0.013 |
| Summer temperature, stream length | 52.006 | 0.011 |
| Pool width, number of deep pools | 53.412 | 0.006 |
| Pool width, number of pools with clean gravel | 54.171 | 0.004 |
| Pool width, number of pools with structure | 54.207 | 0.004 |
| Number of all pools | 55.146 | 0.002 |
| Stream length | 55.636 | 0.002 |
| Pool width | 55.656 | 0.002 |
| Intercepts only | 61.014 | 0.000 |
| Winter temperature | 63.274 | 0.000 |
| *Basin-Scale Habitat* | | |
| Watershed area, basin aspect | 56.173 | 0.201 |
| Watershed area | 56.437 | 0.177 |
| Watershed area, basin relief | 57.029 | 0.131 |

Table 2.5. Continued.

| | | |
|---|---|---|
| Watershed area, latitude | 58.046 | 0.079 |
| Watershed area, basin relief, latitude | 58.806 | 0.054 |
| Watershed area, elevation | 58.894 | 0.052 |
| Main channel length | 58.907 | 0.051 |
| Watershed area, main channel length | 59.086 | 0.047 |
| Watershed area, basin slope | 59.124 | 0.046 |
| Watershed area, basin aspect, elevation | 59.165 | 0.045 |
| Watershed area, elevation, latitude | 60.805 | 0.020 |
| Main channel length, elevation | 60.959 | 0.018 |
| Intercepts only | 61.014 | 0.018 |
| Main channel length, stream slope | 61.531 | 0.014 |
| Main channel length, stream aspect | 61.655 | 0.013 |
| Watershed area, elevation, latitude, elevation*latitude | 62.121 | 0.010 |
| Main channel length, elevation, latitude | 62.392 | 0.009 |
| Drainage density | 63.143 | 0.006 |
| Main channel length, stream aspect, elevation | 63.951 | 0.004 |
| Main channel length, elevation, latitude, elevation*latitude | 64.102 | 0.004 |

Table 2.5. Concluded.

| | | |
|---|---|---|
| *Stream- and Basin-Scale Habitat* | | |
| Summer temperature, pool width, deep pools | 45.518 | 0.366 |
| Summer temperature, pool width | 46.678 | 0.205 |
| Summer temperature, width, deep pools, watershed area, basin aspect | 47.005 | 0.174 |
| Summer temperature, pool width, watershed area | 48.420 | 0.086 |
| Summer temperature, width, deep pools, watershed area | 48.847 | 0.069 |
| Summer temperature, watershed area | 50.816 | 0.026 |
| Summer temperature, deep pools, watershed area | 51.240 | 0.021 |
| Pool width, deep pools, watershed area, basin aspect | 51.557 | 0.018 |
| Summer temperature | 51.656 | 0.017 |
| Pool width, watershed area, basin aspect | 53.298 | 0.007 |
| Pool width, watershed area | 54.145 | 0.005 |
| Pool width, deep pools, watershed area | 54.992 | 0.003 |
| Watershed area | 56.437 | 0.002 |
| Watershed area, basin aspect | 56.173 | 0.002 |
| Intercepts only | 61.014 | 0.000 |

Table 2.6. Maximum likelihood estimates from ordinal polytomous logistic regression of intercept and slope parameters for the four "best approximating" models predicting cutthroat trout translocation success from water temperature and stream-scale habitat collected during field surveys. Standard errors are in parentheses. All models are significant at $P = 0.0001$.

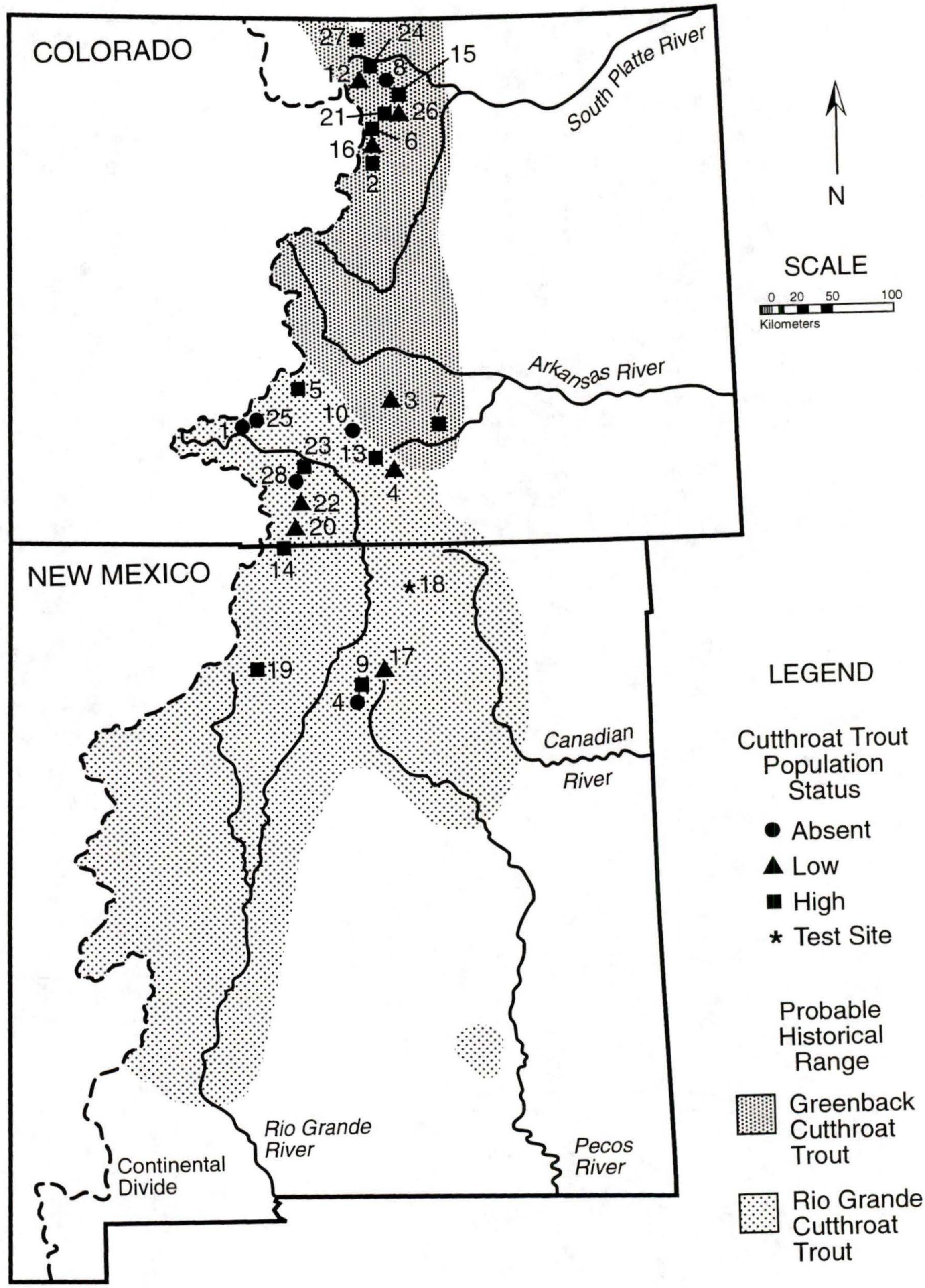
| Model | Intercept 1 | Intercept 2 | Mean July temperature | Pool width | Number of pools[a] |
|---|---|---|---|---|---|
| Summer temperature, pool width, deep pools[a] | 11.454 | 14.077 | -0.891 | -1.451 | -0.017 |
| | (4.460) | (4.868) | (0.355) | (0.713) | (0.010) |
| Summer temperature, pool width | 10.345 | 12.688 | -0.889 | -1.460 | - |
| | (3.853) | (4.179) | (0.336) | (0.646) | - |
| Summer temperature, pool width, all pools[a] | 10.369 | 12.846 | -0.765 | -1.433 | -0.008 |
| | (4.052) | (4.398) | (0.334) | (0.676) | (0.006) |
| Summer temperature, pool width, pools with structure[a] | 10.277 | 12.749 | -0.816 | -1.352 | -0.012 |
| | (3.970) | (4.327) | (0.330) | (0.668) | (0.009) |

[a] Three models included number of pools that met specific criteria (see text and Appendix III).

Table 2.7. Maximum likelihood estimates from ordinal polytomous logistic regression of intercept and slope parameters for the four "best approximating" models predicting cutthroat trout translocation success from basin-scale habitat measured from digital data (see Appendix IV). Standard errors are in parentheses. All models are significant at $P \leq 0.016$.

| Model | Intercept 1 | Intercept 2 | Watershed area | Basin aspect | Basin relief | Latitude |
|---|---|---|---|---|---|---|
| Watershed area, basin aspect | -0.867 | 0.829 | -0.144 | 0.039 | - | - |
| | (1.164) | (1.145) | (0.081) | (0.024) | - | - |
| Watershed area | 0.251 | 1.804 | -0.123 | - | - | - |
| | (0.869) | (0.916) | (0.069) | - | - | - |
| Watershed area, basin relief | -1.177 | 0.478 | -0.176 | - | 0.002 | - |
| | (1.366) | (1.337) | (0.096) | - | (0.002) | - |
| Watershed area, latitude | 9.901 | 11.529 | -0.125 | - | - | -0.000002 |
| | (9.184) | (9.265) | (0.068) | - | - | (0.000002) |

Figure 2.1. The approximate historical range of greenback and Rio Grande cutthroat trout (adapted from Behnke 1992; Stumpff and Cooper 1996) and location of the 28 study streams where cutthroat trout were translocated. Stream numbers correspond to those in Table 2.1.



COLORADO
NEW MEXICO
South Platte River
Arkansas River
Canadian River
Rio Grande River
Pecos River
Continental Divide
N
SCALE
0 20 50 100
Kilometers
LEGEND
Cutthroat Trout Population Status
Absent
Low
High
Test Site
Probable Historical Range
Greenback Cutthroat Trout
Rio Grande Cutthroat Trout

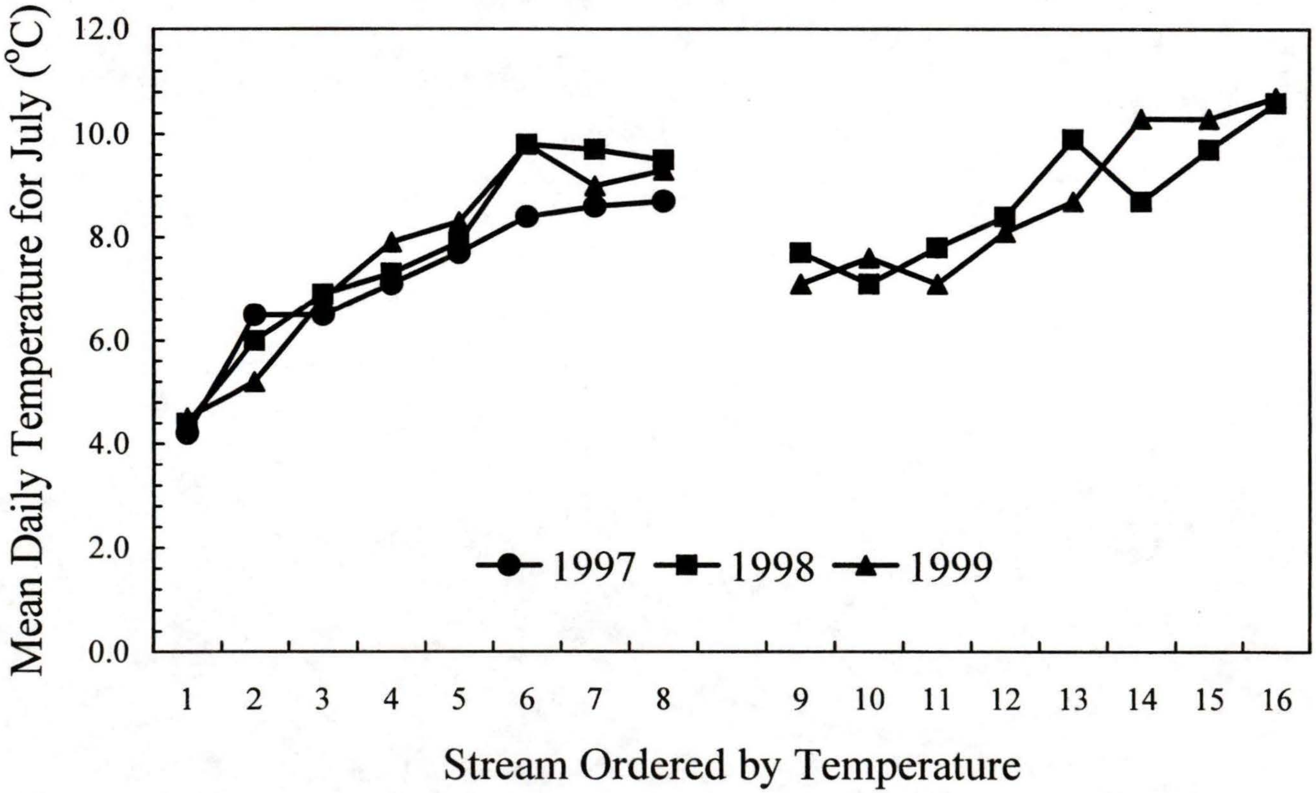



Figure 2.2. Mean daily water temperature (degrees C) for July for the 16 study streams with two or more years of data, ordered by temperature. There was no significant difference in July temperatures among years ($P$ = 0.15 by ANOVA).

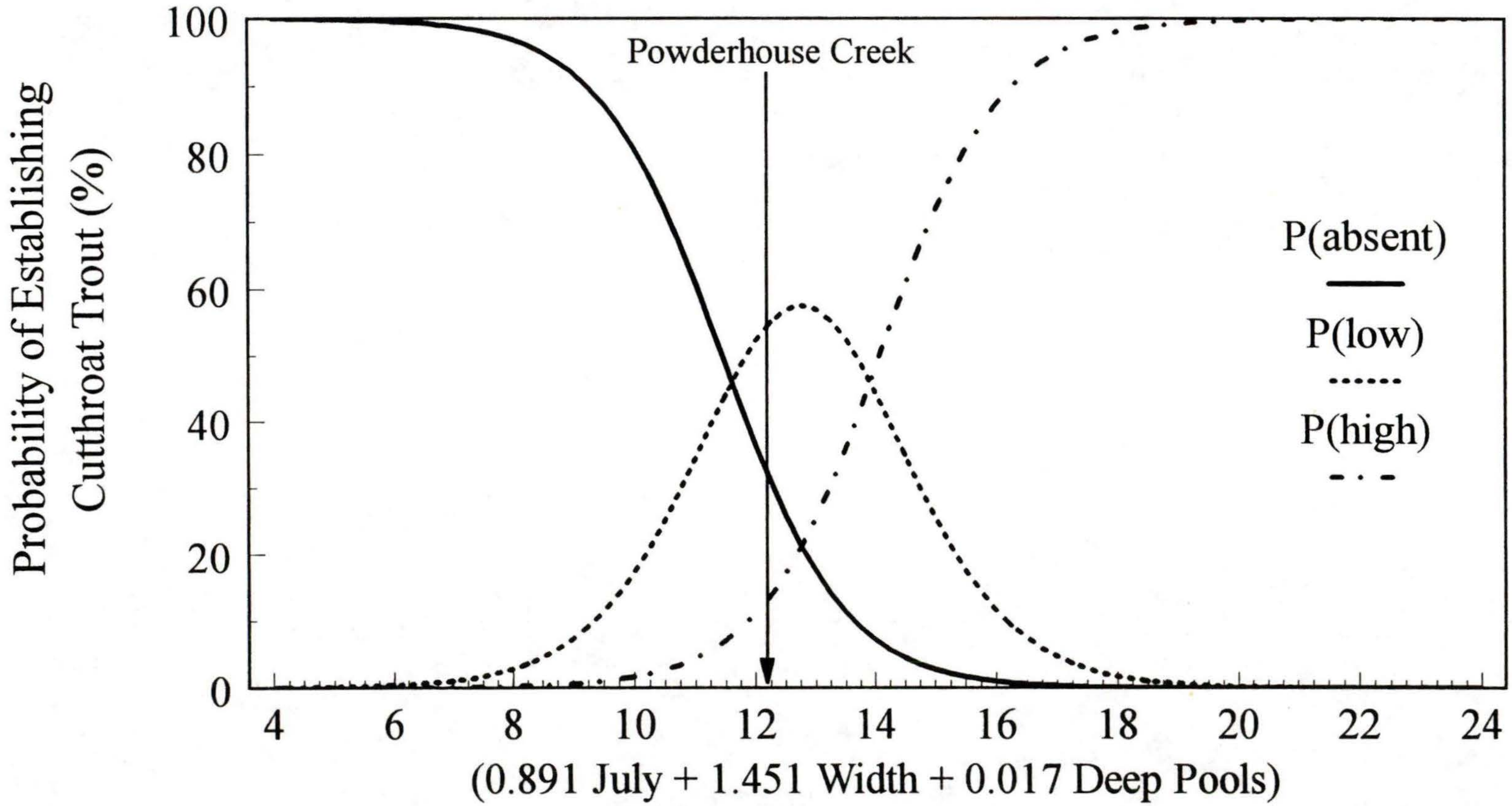



Figure 2.3. The "best" model for predicting success of translocated cutthroat trout populations from stream-scale habitat attributes. Curves show the predicted probability of translocation success based on a polytomous logistic regression function (shown on the abscissa) including mean daily water temperature for July (degrees C), mean pool bankfull width (m), and total number of deep pools (residual depth >30 cm). This model can be used to estimate the probability that a future translocation site will be successful. For example, predicted probabilities for Powderhouse Creek were only 13% for supporting high numbers of cutthroat trout, 54% for supporting low numbers, and 33% for supporting no cutthroat trout (shown by the arrow).

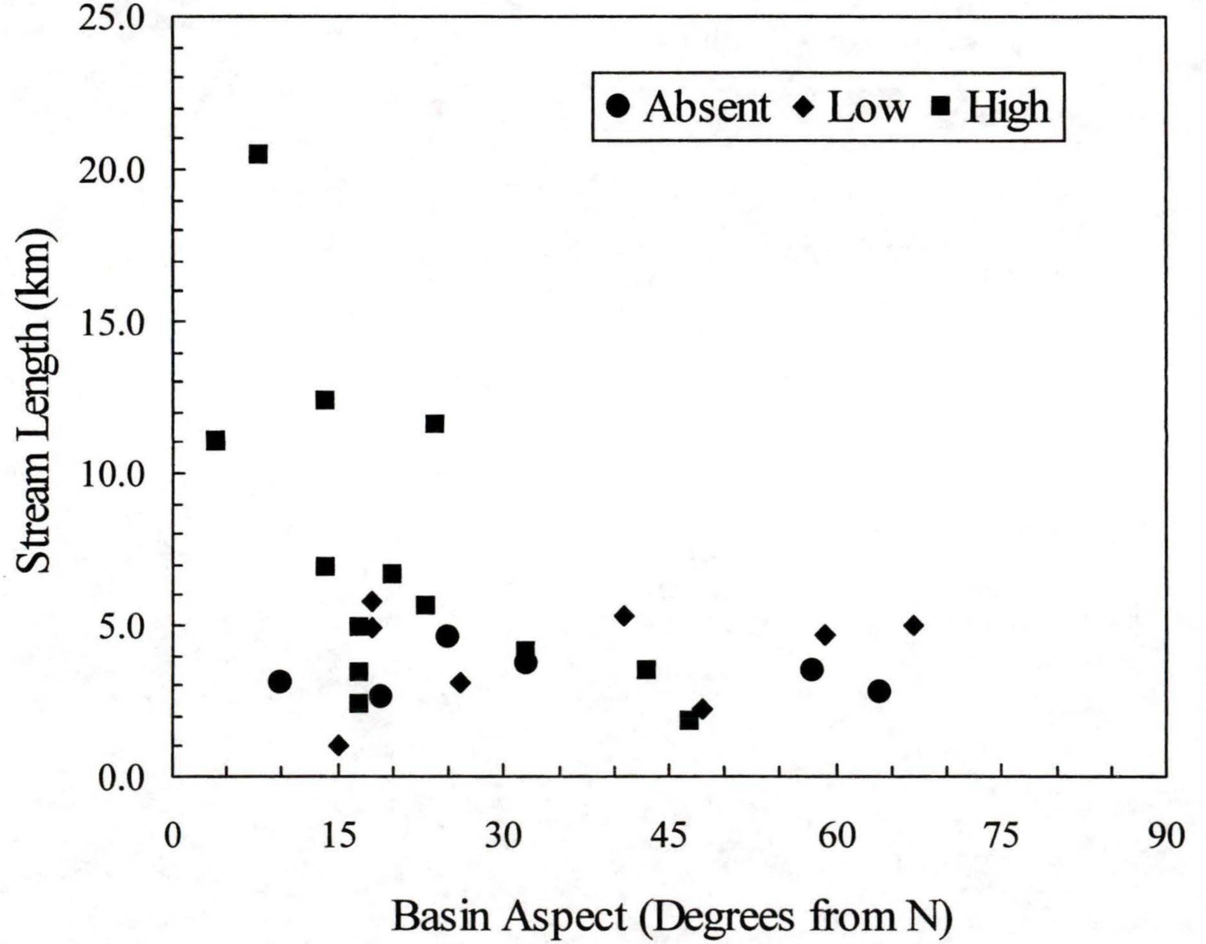



Figure 2.4. Basin aspect versus stream length (km), measured as the distance from the downstream fish movement barrier to the upstream end of pool habitat (bankfull width < 2.0 m). Symbols show population status of the translocation streams. Four large basins with high numbers of cutthroat trout that trend North-South likely caused spurious statistical significance for basin aspect in the logistic regression model.

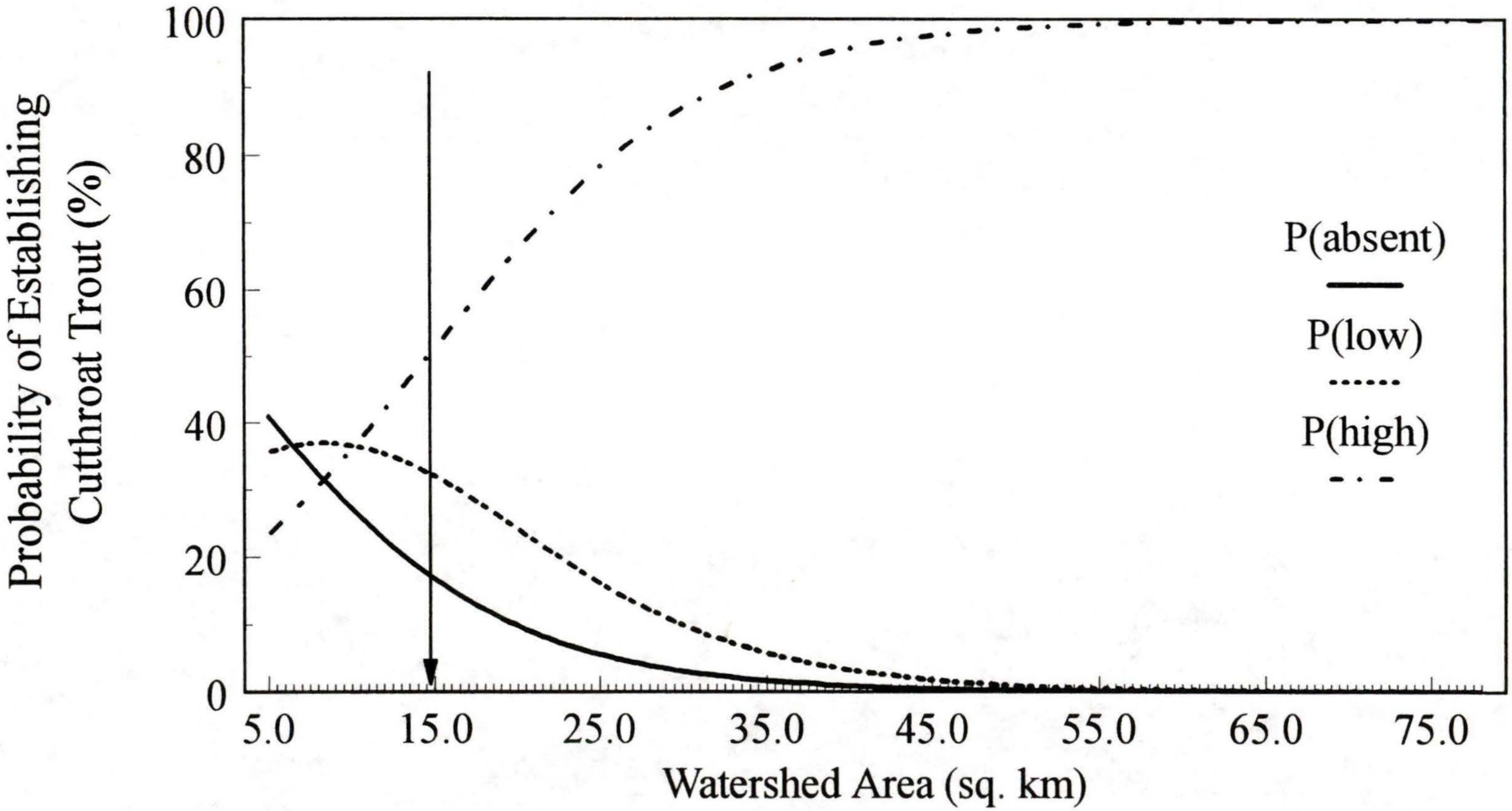



Figure 2.5. The "best" model for predicting success of translocated cutthroat trout populations from basin-scale habitat attributes. Curves show the predicted probability of translocation success based on a polytomous logistic regression function of watershed area (square kilometers). Translocations have greater than a 50% chance of establishing high numbers of cutthroat trout in watersheds greater than 14.7 square kilometers (shown by the arrow) .

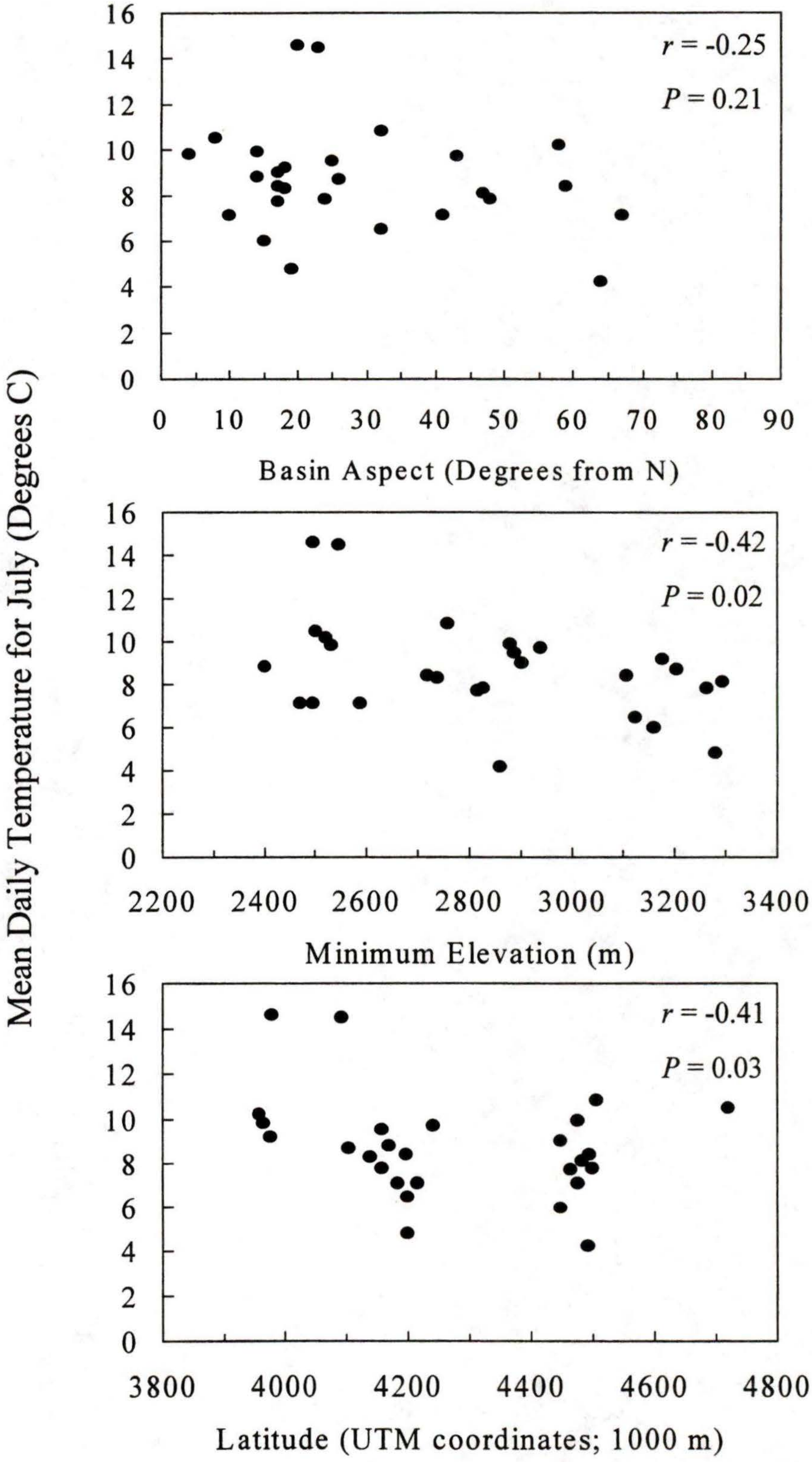



Figure 2.6. The relationship between aspect (degrees from N), elevation (m), and latitude (Universal Transmercator coordinates in 1000 m) and July water temperatures (degrees C) measured with thermographs for 27 cutthroat trout translocation streams.

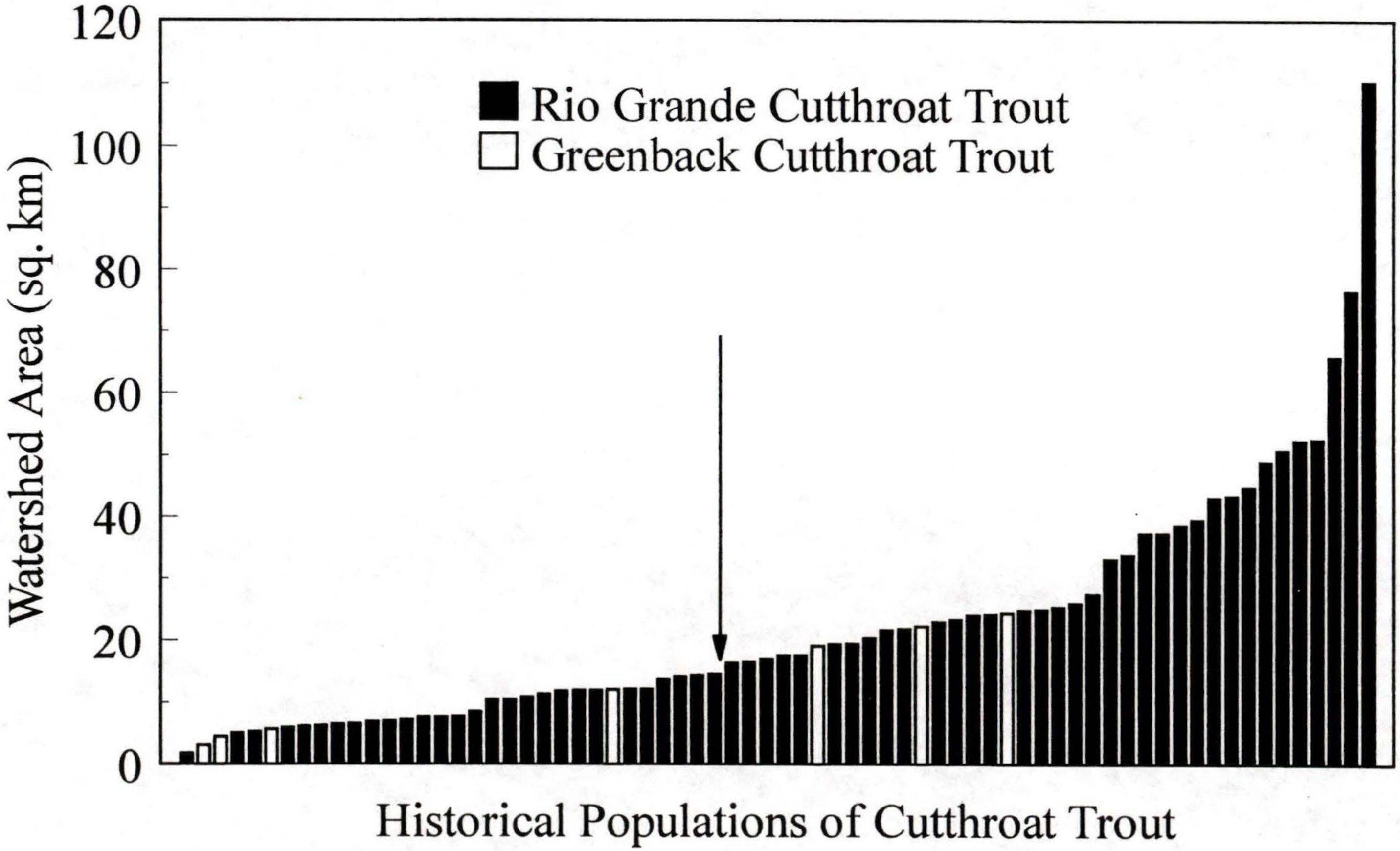



Figure 2.7. The distribution of watershed area (square kilometers) for 70 historical stream populations of greenback and Rio Grande cutthroat trout. Thirty-two populations in watersheds < 14.7 square kilometers (shown by the arrow) may be at greater risk from extirpation due to insufficient habitat (see text).

APPENDIX I:

GREENBACK CUTTHROAT TROUT TRANSLOCATION PROJECTS

Appendix I. Greenback cutthroat trout translocation projects carried out prior to 1995 (USFWS 1998 and personal communications) in the South Platte (SP) and Arkansas (AR) drainage basins. Locations include Arapaho-Roosevelt National Forest (ARNF), Fort Carson Military Base, Leadville National Fish Hatchery (LNFH), Rocky Mountain National Park (RMNP), Pike-San Isabel National Forest (PSINF), and city water sources. Chemical treatment is the date of treatment with rotenone or antimycin to remove nonnative salmonids prior to introduction of cutthroat trout. No chemical treatment indicates that the habitat was barren of fish, and netted indicates that nonnative salmonids were removed by gillnet prior to introduction. Initial stocking includes the source of fish (W = wild, H = hatchery) and the year of first introduction.

| No. | Greenback cutthroat trout translocation water | Drainage | Location | Stream (km) | Lake (ha) | Median elevation (m) | Potential barrier to fish movement | Chemical treatment | Initial stocking |
|---|---|---|---|---|---|---|---|---|---|
| *Successful Translocations* | | | | | | | | | |
| 1 | North Fork Big Thompson, Lake Louise, and ponds | SP | RMNP | 1.7 | 13.1 | 3320 | Waterfall | None | W: 1971 |
| 2 | Bear Lake | SP | RMNP | 0.1 | 4.5 | 2890 | Steep cascades | 1975 | W: 1975 |
| 3 | Williams Gulch | SP | ARNF | 2.0 | 0.0 | 2900 | Steep cascades | None | H: 1981 |
| 4 | Sheep Creek | SP | ARNF | 11.2 | 0.0 | 2970 | Steep cascades | 1981 | H: 1982 |
| 5 | Fern Lake and Fern Creek | SP | RMNP | 1.4 | 3.7 | 2850 | Fern Falls | 1982 | H: 1982 |
| 6 | Odessa Lake | SP | RMNP | 0.2 | 4.5 | 3030 | Steep cascades | None | H: 1984 |
| 7 | Lawn Lake | SP | RMNP | 0.9 | 9.5 | 3350 | Steep cascades | 1983 | H: 1984 |
| 8 | Roaring River | SP | RMNP | 6.5 | 0.0 | 3080 | Horseshoe Falls | 1983 | H: 1984 |

Appendix I. Continued.

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| 9 | Lower Hutcheson Lake | SP | RMNP | 1.0 | 1.7 | 3260 | Steep cascades | 1987 | H: 1989 |
| 10 | Pear Lake | SP | RMNP | 1.2 | 6.1 | 3200 | Steep cascades | 1988 | H: 1989 |
| 11 | Sandbeach Lake | SP | RMNP | 0.1 | 4.0 | 3140 | Steep cascades | 1988 | H: 1989 |
| 12 | Spruce Lake | SP | RMNP | 0.3 | 1.5 | 2940 | Steep cascades | None | H: 1991 |
| 13 | Cony Creek | SP | RMNP | 3.5 | 0.0 | 3040 | Calypso Cascades | 1988 | H: 1989 |
| 14 | Boehmer Reservoir | AR | Colorado Springs | 0.0 | 10.1 | 3500 | Reservoir spillway Height = 18 m | 1984 | H: 1985 |
| *Unsuccessful Translocations for Reasons Apparently Related to Habitat Size or Quality* | | | | | | | | | |
| 15 | May Creek | SP | ARNF | 1.7 | 0.0 | 3040 | Steep cascades | None | W: 1980 |
| 16 | Loomis Lake | SP | RMNP | 0.4 | 1.5 | 3120 | Steep cascades | None | H: 1991 |
| 17 | West Creek | SP | RMNP | 2.4 | 0.0 | 2960 | West Falls | 1978 | W: 1979 |
| 18 | Ouzel Creek (below Bluebird Lake) | SP | RMNP | 1.4 | 0.0 | 3180 | Steep cascades | 1980 | H: 1981 |
| 19 | Big Crystal Lake | SP | RMNP | 0.0 | 10.0 | 3510 | Waterfall | None | H: 1984 |
| 20 | Hourglass Creek | SP | ARNF | 2.0 | 0.0 | 2990 | Gage station and fish weir | None | W: 1965 |

Appendix I. Continued.

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| 21 | Lily Lake | SP | RMNP | 0.0 | 5.0 | 2610 | Cement dam | Netted | H: 1986 |
| 22 | North Fork Jackson Creek | SP | PSINF | 0.3 | 0.0 | 2950 | Intermittent channel | 1984 | H: 1987 |
| 23 | Cottonwood Creek | AR | PSINF | 6.4 | 0.0 | 3150 | Dry channel | None | H: 1983 |
| 24 | Greenhorn Creek | AR | PSINF | 3.2 | 0.0 | 3270 | Waterfalls | None | H: 1988 |
| | *Unsuccessful Translocations Due to Reinvasion by Nonnative Salmonids* | | | | | | | | |
| 25 | George Creek | SP | ARNF | 12.7 | 0.0 | 2570 | Rock gabion Height = 1 - 2 m | 1982 | H: 1983 |
| 26 | Black Hollow Creek | SP | ARNF | 5.2 | 0.0 | 2540 | Wooden structure | 1967, 1979 | W: 1967 |
| 27 | Bruno Gulch | SP | PSINF | 9.0 | 1.4 | 3060 | Wooden structure Height = 2.4 m | 1985 | H: 1987 |
| 28 | Pennock Creek | SP | ARNF | 8.0 | 0.0 | 2760 | Rock gabion Height = 1.4 m | 1985 | H: 1986 |
| 29 | Cornelius Creek | SP | ARNF | 6.9 | 0.0 | 2500 | Rock gabion Height = 1 - 2 m | 1982 | H: 1983 |
| 30 | Ouzel Lake and Ouzel Creek (above Ouzel Falls) | SP | RMNP | 4.7 | 2.6 | 3000 | Ouzel Falls | 1980 | H: 1981 |

Appendix I. Concluded.

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| 31 | Hidden Valley Creek and beaver ponds | SP | RMNP | 1.6 | 2.5 | 2760 | Steep cascades | 1973 | H: 1973 |
| 32 | Lake Fork complex | AR | PSINF | 4.4 | 11.3 | 3290 | Wooden structure Height = 2 - 3 m | 1986 | H: 1987 |
| 33 | Rock Creek complex | AR | LNFH, PSINF | 10.3 | 8.3 | 3290 | Water diversion | 1990 | H: 1991 |
| 34 | Husted Lake | SP | RMNP | 0.1 | 4.1 | 3380 | Steep cascades | 1986, 1990 | H: 1991 |
| 35 | North Fork Big Thompson River and Lost Lake | SP | RMNP | 3.0 | 3.7 | 3140 | Lost Falls | 1986 | H: 1987 |

*Unsuccessful Translocations Due to Metal Pollution (36) and Bird Predation (37)*

| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| 36 | Bard Creek | SP | ARNF | 6.1 | 0.0 | 2960 | Steep cascades | None | H: 1982 |
| 37 | Lytle and Duck Pond | AR | Fort Carson | 1.0 | 2.1 | 1890 | No outlet stream | None | W: 1981 |

APPENDIX II:

AGENCY STOCKING RECORDS FOR GREENBACK AND RIO GRANDE CUTTHROAT TROUT TRANSLOCATION STREAMS

Appendix II. Unpublished data from natural resource agencies on the stocking frequency, source and number of fish, and approximate size of individuals translocated to the 28 greenback (GB) or Rio Grande (RG) cutthroat trout study streams (Harig and Fausch 1996; Y. Paroz, New Mexico Department of Game and Fish, personal communication; unpublished reports from the Colorado Division of Wildlife and U.S. Fish and Wildlife Service). Dashes indicate that data were unavailable.

| Stream | Sub-species | Number of stocking events | Source of fish | Approximate number of fish stocked | Approximate size of individuals (cm) |
|---|---|---|---|---|---|
| *Absent* | | | | | |
| Benito Creek | RG | 2 | Wild | 275 | 8 – 15 |
| Doctor Creek | RG | Natural[a] | Wild | -- | -- |
| Hourglass Creek | GB | 4 | Wild and hatchery | 1686 | 3 – 15 |
| Little Medano Creek | RG | 2 | Wild | 400 | 10 – 15 |
| Unknown Creek | RG | 1 | Wild | 130 | 8 |
| W.F. San Francisco Creek | RG | 1 | Wild | 65 | 13 – 15 |
| *Low Population* | | | | | |
| Cottonwood Creek | GB | 7 | -- | -- | -- |
| Little Ute Creek | RG | 3 | Wild | 522 | 2 – 15 |
| May Creek | GB | 5 | Wild and hatchery | 6255 | 2 – 15 |
| Ouzel Creek | GB | 3 | Hatchery | -- | 2 – 8 |
| Pecos River | RG | 2 | Wild | 63 | -- |
| Rio de los Pinos | RG | 1 | Wild | 300 | 5 – 25 |

Appendix II. Concluded.

| | | | | | |
|---|---|---|---|---|---|
| Rough Canyon/ Rhodes Gulch | RG | 1 | Wild | 230 | 8 |
| West Creek | GB | 4 | Wild and hatchery | 4158 | 2 – 15 |
| *High Population* | | | | | |
| Cony Creek | GB | 1 | Hatchery | 5000 | 2 – 5 |
| East Middle Creek | RG | 1 | Wild | 515 | 10 – 15 |
| Fern Creek | GB | 3 | Hatchery | >36,000 | 2 – 5 |
| Greenhorn Creek | GB | 1 | -- | -- | -- |
| Jacks Creek | RG | 2 | Wild | 219 | -- |
| Medano Creek | RG | 3 | Wild and broodstock | 3152 | 13 |
| Nabor Creek | RG | 1 | Wild | 736 | 8 – 15 |
| N.F. Big Thompson River | GB | 2 | Wild and hatchery | 4000 | 3 – 13 |
| Rio Cebolla | RG | 2 | Wild | 865 | -- |
| Roaring River | GB | 4 | Hatchery | >20,000 | 2 – 5 |
| San Francisco Creek | RG | 2 | Wild | 182 | 8 – 15 |
| Sheep Creek | GB | 5 | Hatchery | >15,000 | 3 – 10 |
| Williams Gulch | GB | 4 | Hatchery | 4109 | 2 – 15 |
| *Test Translocation Site* | | | | | |
| Powderhouse Creek | RG | 1 | Wild | 340 | -- |

[a] Prior to treatment with rotenone, Rio Grande cutthroat trout in Doctor Creek were moved upstream above an upper barrier to fish movement (number of fish is unknown). It was assumed that these fish would move downstream and repopulate the stream after treatment (Y. Paroz, New Mexico Department of Game and Fish, personal communication). The status of the cutthroat trout above the upper barrier is unknown, but we did not see any in a ~ 100 meter reach just upstream of the barrier during our habitat survey.

# APPENDIX III:

# VALUES FOR REMAINING STREAM-SCALE HABITAT VARIABLES

Appendix III. Values for additional stream-scale habitat variables quantified from field surveys of the 28 study streams where Rio Grande or greenback cutthroat trout were translocated.

| Stream | Number of pools | Number of pools with structure | Number of pools with clean gravel | Mean daily temperature for Dec - Feb (°C) |
|---|---|---|---|---|
| *Absent* | | | | |
| Benito Creek | 51 | 8 | 0 | 1.4 |
| Doctor Creek | 71 | 40 | 34 | 0.2 |
| Hourglass Creek | 154 | 100 | 72 | 0.3 |
| Little Medano Cr. | 47 | 36 | 10 | 0.4 |
| Unknown Creek | 45 | 10 | 6 | 0.8 |
| W.F. San Francisco Cr. | 67 | 33 | 1 | 0.5 |
| Mean (SE) | 73 (17) | 38 (14) | 21 (11) | 0.6 (0.2) |
| *Low Population* | | | | |
| Cottonwood Creek | 192 | 86 | 72 | 1.7 |
| Little Ute Creek | 39 | 16 | 20 | 0.2 |
| May Creek | 325 | 174 | 207 | 0.2 |
| Ouzel Creek | 24 | 15 | 7 | 0.6 |
| Pecos River | 148 | 56 | 65 | 0.2 |
| Rio de los Pinos | 118 | 71 | 84 | 0.7 |

Appendix III. Continued.

| | | | | |
|---|---|---|---|---|
| Rough C./ Rhodes G. | 141 | 71 | 64 | 0.2 |
| West Creek | 138 | 114 | 19 | 0.3 |
| Mean (SE) | 141 (33) | 75 (18) | 67 (22) | 0.5 (0.2) |
| | | | | |
| *High Population* | | | | |
| Cony Creek | 227 | 153 | 116 | 1.1 |
| East Middle Creek | 162 | 78 | 155 | 0.2 |
| Fern Creek | 84 | 58 | 47 | 0.2 |
| Greenhorn Creek | 151 | 74 | 27 | 0.3 |
| Jacks Creek | 346 | 217 | 155 | 0.5 |
| Medano Creek | 571 | 313 | 286 | 0.3 |
| Nabor Creek | 215 | 108 | 114 | 0.4 |
| N.F. Big Thompson R. | 37 | 21 | 0 | 0.3 |
| Rio Cebolla | 214 | 44 | 33 | 2.6 |
| Roaring River | 81 | 35 | 51 | 0.0 |
| San Francisco Creek | 217 | 88 | 54 | 0.2 |
| Sheep Creek | 490 | 302 | 218 | 0.0 |
| Williams Gulch | 173 | 118 | 85 | 0.0 |
| Mean (SE) | 228 (43) | 124 (27) | 103 (23) | 0.5 (0.2) |

Appendix III. Concluded.

| | | | | |
|---|---|---|---|---|
| *Test Translocation Site* | | | | |
| Powderhouse Creek | 76 | 42 | 73 | 0.0 |

# APPENDIX IV:

# VALUES FOR REMAINING BASIN-SCALE HABITAT VARIABLES

Appendix IV. Values for additional basin-scale habitat variables measured with a Geographic Information System from digital data for the 28 study streams where Rio Grande or greenback cutthroat trout were translocated.

| Stream | Main channel length (km) | Total stream length (km) | Drainage density ($km/km^2$) | Basin relief (m) | Mean basin slope (%) | Stream aspect (degrees from N) |
|---|---|---|---|---|---|---|
| *Absent* | | | | | | |
| Benito Creek | 2.8 | 5.4 | 1.1 | 864 | 35.9 | 66 |
| Doctor Creek | 5.1 | 5.2 | 0.7 | 1175 | 42.3 | 81 |
| Hourglass Creek | 4.5 | 5.0 | 0.5 | 1006 | 28.0 | 88 |
| Little Medano Cr. | 7.2 | 14.4 | 0.9 | 1479 | 47.0 | 14 |
| Unknown Creek | 4.5 | 4.5 | 0.5 | 627 | 25.4 | 67 |
| W.F. San Francisco Cr. | 7.0 | 7.0 | 1.0 | 985 | 34.2 | 14 |
| Mean (SE) | 5.2 (0.7) | 6.9 (1.5) | 0.8 (0.1) | 1023 (118) | 35.5 (3.4) | 55 (13) |
| *Low Population* | | | | | | |
| Cottonwood Creek | 11.1 | 11.3 | 1.2 | 1620 | 43.1 | 67 |
| Little Ute Creek | 2.5 | 5.5 | 0.6 | 1094 | 50.0 | 50 |
| May Creek | 6.4 | 9.8 | 0.8 | 885 | 16.9 | 88 |
| Ouzel Creek | 3.0 | 3.4 | 0.5 | 849 | 52.9 | 52 |
| Pecos River | 5.6 | 6.9 | 0.6 | 667 | 23.4 | 1 |
| Rio de los Pinos | 3.4 | 4.7 | 0.5 | 504 | 20.0 | 4 |

Appendix VI. Continued.

| | | | | | | |
|---|---|---|---|---|---|---|
| Rough C./ Rhodes G. | 4.6 | 9.2 | 0.9 | 922 | 22.1 | 80 |
| West Creek | 9.6 | 18.4 | 0.7 | 1584 | 31.6 | 77 |
| Mean (SE) | 5.8 (1.1) | 8.7 (1.7) | 0.7 (0.1) | 1016 (142) | 32.5 (5.0) | 52 (12) |
| *High Population* | | | | | | |
| Cony Creek | 7.4 | 9.2 | 0.6 | 1103 | 37.6 | 75 |
| East Middle Creek | 5.2 | 9.3 | 0.7 | 1095 | 34.6 | 53 |
| Fern Creek | 4.3 | 4.7 | 0.7 | 942 | 50.1 | 84 |
| Greenhorn Creek | 2.7 | 4.7 | 0.7 | 571 | 19.0 | 67 |
| Jacks Creek | 11.2 | 11.2 | 0.6 | 1290 | 26.2 | 1 |
| Medano Creek | 20.2 | 67.8 | 0.9 | 1565 | 43.6 | 16 |
| Nabor Creek | 6.3 | 8.0 | 0.7 | 916 | 27.5 | 23 |
| N.F. Big Thompson R. | 2.0 | 5.2 | 0.6 | 854 | 39.0 | 89 |
| Rio Cebolla | 14.9 | 29.1 | 0.8 | 638 | 22.5 | 35 |
| Roaring River | 6.9 | 8.7 | 0.6 | 1279 | 45.3 | 22 |
| San Francisco Creek | 27.0 | 40.6 | 1.0 | 1629 | 27.6 | 30 |
| Sheep Creek | 15.5 | 25.9 | 0.7 | 764 | 17.4 | 39 |
| Williams Gulch | 5.1 | 6.7 | 0.7 | 339 | 12.1 | 60 |
| Mean (SE) | 9.9 (2.1) | 17.8 (5.2) | 0.7 (0.0) | 999 (105) | 31.0 (3.3) | 46 (8) |

Appendix VI. Concluded.

| | | | | | | |
|---|---|---|---|---|---|---|
| *Test Translocation Site* | | | | | | |
| Powderhouse Creek | 5.0 | 5.5 | 0.6 | 910 | 28.6 | 74 |

# APPENDIX V:

# VALUES FOR BASIN-SCALE HABITAT VARIABLES FOR HISTORICAL STREAM POPULATIONS OF GREENBACK AND RIO GRANDE CUTTHROAT TROUT

Appendix V. Values for basin-scale habitat variables measured with a Geographic Information System from digital data for 70 historical stream populations of greenback and Rio Grande cutthroat trout.

| Stream | Latitude | Longitude | Watershed area ($km^2$) | Main channel length (km) | Total stream length (km) | Drainage density ($km/km^2$) | Basin relief (m) | Minimum elevation (m) | Mean basin slope (%) | Mean channel slope (%) | Basin aspect (degrees from N) | Stream aspect (degrees from N) |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| *Greenback Cutthroat Trout Populations* | | | | | | | | | | | | |
| Cascade Cr | 37° 38′ 52″ | 105° 24′ 07″ | 4.4 | 2.1 | 2.3 | 0.5 | 697 | 3068 | 33.2 | 16.8 | 48 | 75 |
| Como Cr | 40° 00′ 59″ | 105° 30′ 51″ | 5.6 | 2.9 | 2.9 | 0.5 | 469 | 2695 | 15.9 | 8.6 | 63 | 53 |
| Hague Cr | 40° 28′ 57″ | 105° 40′ 22″ | 3.0 | 2.3 | 2.8 | 1.0 | 756 | 3377 | 53.2 | 12.9 | 0 | 72 |
| Hunters Cr | 40° 12′ 47″ | 105° 34′ 21″ | 12.0 | 6.9 | 8.0 | 0.7 | 1463 | 2865 | 42.8 | 12.2 | 21 | 51 |
| Poudre River | 40° 32′ 54″ | 105° 36′ 29″ | 22.2 | 7.5 | 13.8 | 0.6 | 1322 | 2782 | 35.0 | 16.7 | 21 | 16 |
| Roaring Cr | 40° 43′ 11″ | 105° 44′ 27″ | 24.3 | 11.1 | 15.8 | 0.6 | 819 | 2532 | 22.9 | 10.7 | 2 | 7 |
| South Apache Cr | 37° 50′ 26″ | 104° 55′ 44″ | 19.0 | 11.4 | 17.4 | 0.9 | 1619 | 2084 | 49.2 | 18.1 | 35 | 55 |
| *Rio Grande Cutthroat Trout Populations in Colorado* | | | | | | | | | | | | |
| Bennett Cr | 37° 50′ 03″ | 107° 07′ 55″ | 11.9 | 7.4 | 9.1 | 0.8 | 882 | 2914 | 20.2 | 11.6 | 31 | 50 |
| Cascade Cr | 37° 00′ 52″ | 106° 21′ 41″ | 6.4 | 3.9 | 4.6 | 0.7 | 348 | 2984 | 14.5 | 8.7 | 19 | 32 |
| Cat Cr | 37° 23′ 06″ | 106° 14′ 21″ | 76.6 | 20.1 | 65.6 | 0.9 | 1123 | 2517 | 27.4 | 13.2 | 20 | 28 |
| Cuates Cr | 37° 01′ 32″ | 105° 22′ 51″ | 11.9 | 8.1 | 9.5 | 0.8 | 1245 | 2644 | 36.9 | 15.1 | 31 | 83 |

Appendix V.  Continued.

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| East Pass Cr | 38° 09′ 55″ | 106° 27′ 21″ | 37.3 | 12.3 | 28.9 | 0.8 | 755 | 2597 | 21.5 | 7.5 | 39 | 45 |
| Jaroso Cr | 37° 02′ 59″ | 105° 22′ 28″ | 21.8 | 11.8 | 20.0 | 0.9 | 1270 | 2642 | 30.2 | 11.8 | 13 | 57 |
| Johns Cr & Bear Cr | 38° 00′ 29″ | 106° 39′ 24″ | 38.5 | 18.6 | 24.4 | 0.6 | 1089 | 2855 | 25.2 | 8.9 | 18 | 9 |
| Middle Fork Carnero Cr | 37° 54′ 31″ | 106° 23′ 49″ | 48.9 | 16.4 | 38.0 | 0.8 | 864 | 2639 | 22.7 | 8.0 | 42 | 51 |
| North Fork Carnero Cr | 37° 54′ 21″ | 106° 23′ 24″ | 65.9 | 13.7 | 49.7 | 0.8 | 788 | 2630 | 22.6 | 8.4 | 7 | 17 |
| Osier Cr | 37° 05′ 19″ | 106° 18′ 11″ | 33.1 | 9.1 | 19.6 | 0.6 | 669 | 2615 | 29.3 | 10.0 | 24 | 42 |
| Torcido Cr | 37° 04′ 28″ | 105° 21′ 16″ | 16.3 | 9.2 | 12.4 | 0.8 | 936 | 2653 | 30.2 | 11.7 | 20 | 57 |
| Willow Cr | 37° 01′ 09″ | 105° 22′ 52″ | 21.6 | 8.9 | 17.3 | 0.8 | 1275 | 2643 | 34.5 | 16.0 | 46 | 79 |
| Wolf Cr | 37° 01′ 18″ | 106° 27′ 57″ | 10.4 | 2.7 | 7.7 | 0.7 | 544 | 2960 | 21.4 | 14.5 | 47 | 40 |

*Rio Grande Cutthroat Trout Populations in New Mexico*

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Agua Piedra Cr | 36° 08′ 01″ | 105° 31′ 35″ | 16.9 | 6.9 | 14.8 | 0.9 | 1235 | 2575 | 29.1 | 14.5 | 52 | 85 |
| Alamitos Cr | 36° 03′ 03″ | 105° 28′ 20″ | 17.5 | 6.6 | 11.3 | 0.6 | 856 | 2962 | 20.6 | 9.2 | 51 | 89 |
| Rito Angostura | 36° 06′ 24″ | 105° 29′ 11″ | 25.3 | 12.1 | 22.2 | 0.9 | 1221 | 2698 | 26.7 | 12.1 | 49 | 73 |
| Rito Azul | 35° 56′ 58″ | 105° 38′ 24″ | 6.1 | 3.4 | 3.4 | 0.6 | 541 | 3109 | 23.6 | 14.4 | 76 | 71 |
| Bitter Cr | 36° 42′ 22″ | 105° 24′ 03″ | 27.4 | 13.9 | 18.9 | 0.7 | 770 | 2642 | 31.2 | 9.8 | 14 | 22 |
| Cabresto Cr | 36° 44′ 05″ | 105° 30′ 27″ | 52.5 | 18.5 | 34.4 | 0.7 | 1085 | 2562 | 38.5 | 13.8 | 12 | 46 |

Appendix V. Continued.

| | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Rito Café | 36° 01′ 20″ 106° 42′ 26″ | 16.4 | 6.5 | 12.2 | 0.7 | 427 | 2657 | 16.2 | 7.6 | 26 | 38 |
| Rio Capulin | 35° 52′ 04″ 105° 47′ 26″ | 10.8 | 7.7 | 8.2 | 0.8 | 1086 | 2750 | 37.1 | 17.3 | 42 | 65 |
| Canjilon Cr | 36° 32′ 07″ 106° 19′ 08″ | 13.6 | 7.6 | 9.2 | 0.7 | 534 | 2800 | 16.4 | 8.7 | 23 | 8 |
| Canones Cr | 36° 06′ 06″ 106° 30′ 04″ | 37.3 | 12.3 | 27.4 | 0.7 | 715 | 2513 | 22.7 | 9.5 | 8 | 27 |
| Chihuahuenos Cr | 36° 07′ 00″ 106° 28′ 00″ | 20.3 | 13.6 | 18.7 | 0.9 | 669 | 2523 | 26.4 | 9.6 | 6 | 26 |
| Rito de los Chimayosos | 35° 57′ 23″ 105° 36′ 56″ | 7.0 | 1.5 | 3.2 | 0.5 | 710 | 3277 | 33.9 | 11.4 | 30 | 48 |
| Rio Chiquito | 36° 18′ 47″ 105° 25′ 02″ | 24.0 | 9.7 | 14.2 | 0.6 | 739 | 2731 | 27.6 | 11.1 | 6 | 18 |
| Columbine Cr | 36° 40′ 34″ 105° 30′ 54″ | 43.1 | 20.5 | 30.9 | 0.7 | 1456 | 2414 | 47.7 | 19.4 | 2 | 33 |
| Comanche Cr | 36° 49′ 53″ 105° 19′ 03″ | 110.3 | 16.4 | 87.4 | 0.8 | 1093 | 2728 | 23.7 | 7.7 | 6 | 14 |
| Dalton Canyon | 35° 40′ 39″ 105° 45′ 20″ | 12.1 | 5.9 | 9.2 | 0.8 | 752 | 2401 | 40.8 | 14.8 | 13 | 33 |
| El Rito | 36° 32′ 52″ 106° 16′ 45″ | 24.8 | 8.3 | 20.8 | 0.8 | 424 | 2831 | 16.6 | 9.1 | 43 | 37 |
| Frijoles Cr | 36° 15′ 52″ 105° 24′ 00″ | 6.9 | 3.2 | 4.1 | 0.6 | 630 | 3002 | 26.7 | 14.1 | 13 | 7 |
| Gavilan Canyon | 36° 35′ 42″ 105° 28′ 42″ | 7.7 | 4.8 | 5.3 | 0.7 | 985 | 2738 | 50.7 | 24.1 | 6 | 1 |
| Indian Cr | 35° 44′ 07″ 105° 42′ 41″ | 7.2 | 5.3 | 7.2 | 1.0 | 947 | 2523 | 34.1 | 12.4 | 44 | 47 |
| Indian Cr | 36° 10′ 22″ 105° 36′ 47″ | 1.7 | 1.8 | 1.8 | 1.1 | 478 | 2362 | 21.0 | 13.6 | 76 | 48 |
| La Cueva Canyon | 36° 50′ 24″ 105° 18′ 32″ | 8.5 | 4.2 | 7.0 | 0.8 | 763 | 2751 | 30.2 | 11.9 | 32 | 19 |
| La Jara Cr | 36° 07′ 24″ 106° 54′ 56″ | 14.1 | 6.3 | 13.0 | 0.9 | 812 | 2399 | 28.9 | 16.8 | 39 | 52 |
| Little Blue Cr | 36° 14′ 04″ 105° 14′ 10″ | 10.4 | 5.5 | 7.8 | 0.7 | 893 | 2482 | 33.0 | 11.9 | 67 | 64 |

Appendix V. Continued.

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Little Costilla Cr | 36° 47′ 47″ | 105° 17′ 46″ | 14.3 | 9.6 | 10.4 | 0.7 | 1034 | 2787 | 27.2 | 10.3 | 38 | 49 |
| Lake Fork Cabresto Cr | 36° 45′ 01″ | 105° 29′ 28″ | 19.4 | 6.6 | 12.0 | 0.6 | 1066 | 2798 | 41.7 | 18.2 | 3 | 17 |
| Macho Canyon | 35° 42′ 21″ | 105° 43′ 56″ | 22.9 | 9.2 | 15.6 | 0.7 | 920 | 2451 | 38.8 | 15.9 | 16 | 34 |
| McCrystal Cr | 36° 47′ 31″ | 105° 07′ 56″ | 24.1 | 14.8 | 19.6 | 0.8 | 1296 | 2539 | 22.3 | 8.6 | 48 | 62 |
| Rio Nambe | 3° 50′ 36″ | 105° 50′ 43″ | 33.8 | 8.6 | 23.1 | 0.7 | 1478 | 2347 | 39.1 | 16.4 | 29 | 52 |
| Rio Nutrias | 36° 49′ 08″ | 106° 13′ 19″ | 5.0 | 4.3 | 4.3 | 0.9 | 367 | 2785 | 20.8 | 8.8 | 1 | 79 |
| Rito de la Olla | 36° 16′ 22″ | 105° 25′ 00″ | 17.5 | 10.6 | 11.7 | 0.7 | 764 | 2868 | 28.3 | 12.2 | 17 | 37 |
| Rito del Padre & Rito Maestas | 35° 55′ 56″ | 105° 35′ 22″ | 12.1 | 7.4 | 7.4 | 0.6 | 750 | 3060 | 19.6 | 11.8 | 18 | 3 |
| Palociento Cr | 36° 15′ 38″ | 105° 26′ 40″ | 6.5 | 3.5 | 5.3 | 0.8 | 874 | 2768 | 45.5 | 22.8 | 3 | 28 |
| Peralta Canyon | 35° 41′ 06″ | 106° 27′ 24″ | 43.4 | 17.4 | 31.5 | 0.7 | 1164 | 1864 | 41.2 | 14.8 | 20 | 24 |
| Rito de los Pinos | 36° 06′ 05″ | 105° 54′ 24″ | 5.9 | 3.7 | 4.6 | 0.8 | 699 | 2477 | 38.0 | 19.3 | 50 | 39 |
| Policarpio Canyon | 36° 08′ 20″ | 105° 27′ 11″ | 7.6 | 3.6 | 4.6 | 0.6 | 463 | 2772 | 20.2 | 12.9 | 39 | 51 |
| Polvadera Cr | 36° 06′ 59″ | 106° 26′ 20″ | 50.8 | 32.4 | 41.5 | 0.8 | 1022 | 2396 | 28.7 | 12.4 | 8 | 13 |
| Rito la Presa | 36° 10′ 05″ | 105° 27′ 07″ | 23.3 | 9.8 | 15.3 | 0.7 | 838 | 2804 | 32.6 | 13.3 | 3 | 12 |
| Rio Puerco | 36° 03′ 19″ | 106° 53′ 07″ | 14.5 | 11.5 | 13.3 | 0.9 | 613 | 2615 | 17.4 | 9.9 | 21 | 22 |
| Rio Quemado | 36° 01′ 00″ | 105° 47′ 29″ | 52.3 | 24.5 | 42.7 | 0.8 | 1654 | 2334 | 37.0 | 15.3 | 19 | 38 |
| Rito Resumidero & Oso Cr | 36° 06′ 03″ | 106° 44′ 16″ | 24.9 | 8.4 | 21.6 | 0.9 | 592 | 2643 | 15.5 | 10.4 | 54 | 61 |

Appendix V. Concluded.

| | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| Saloz Cr | 36° 12′ 19″ | 105° 31′ 16″ | 11.3 | 3.9 | 6.3 | 0.6 | 771 | 2656 | 34.7 | 15.5 | 15 | 53 |
| Sardinas Canyon | 36° 11′ 12″ | 105° 27′ 20″ | 7.6 | 5.1 | 5.1 | 0.7 | 669 | 2973 | 33.9 | 16.9 | 30 | 30 |
| San Cristobal Cr | 36° 37′ 00″ | 105° 36′ 22″ | 11.8 | 6.1 | 8.3 | 0.7 | 1178 | 2509 | 43.3 | 16.9 | 56 | 89 |
| South Fork Rio Hondo | 36° 34′ 24″ | 105° 30′ 28″ | 19.3 | 7.2 | 12.6 | 0.7 | 1356 | 2559 | 54.4 | 25.2 | 15 | 63 |
| Tienditas Cr | 36° 22′ 18″ | 105° 22′ 27″ | 39.5 | 9.1 | 29.9 | 0.8 | 776 | 2541 | 22.3 | 11.3 | 6 | 5 |
| Rio de Truchas | 36° 05′ 09″ | 105° 52′ 19″ | 44.8 | 21.0 | 43.3 | 1.0 | 1477 | 2141 | 19.8 | 7.9 | 18 | 59 |
| Rio Valdez | 35° 56′ 13″ | 105° 32′ 15″ | 5.2 | 3.1 | 3.1 | 0.6 | 345 | 3293 | 14.5 | 9.6 | 49 | 39 |
| Vidal Cr | 36° 45′ 24″ | 105° 16′ 06″ | 25.9 | 8.0 | 15.4 | 0.6 | 532 | 2845 | 17.7 | 4.2 | 32 | 8 |
| Yerba Cr | 36° 34′ 11″ | 105° 31′ 04″ | 6.2 | 3.7 | 4.3 | 0.7 | 1083 | 2522 | 46.9 | 21.2 | 45 | 40 |

APPENDIX VI:

THERMOGRAPH DATA FROM 1998-1999

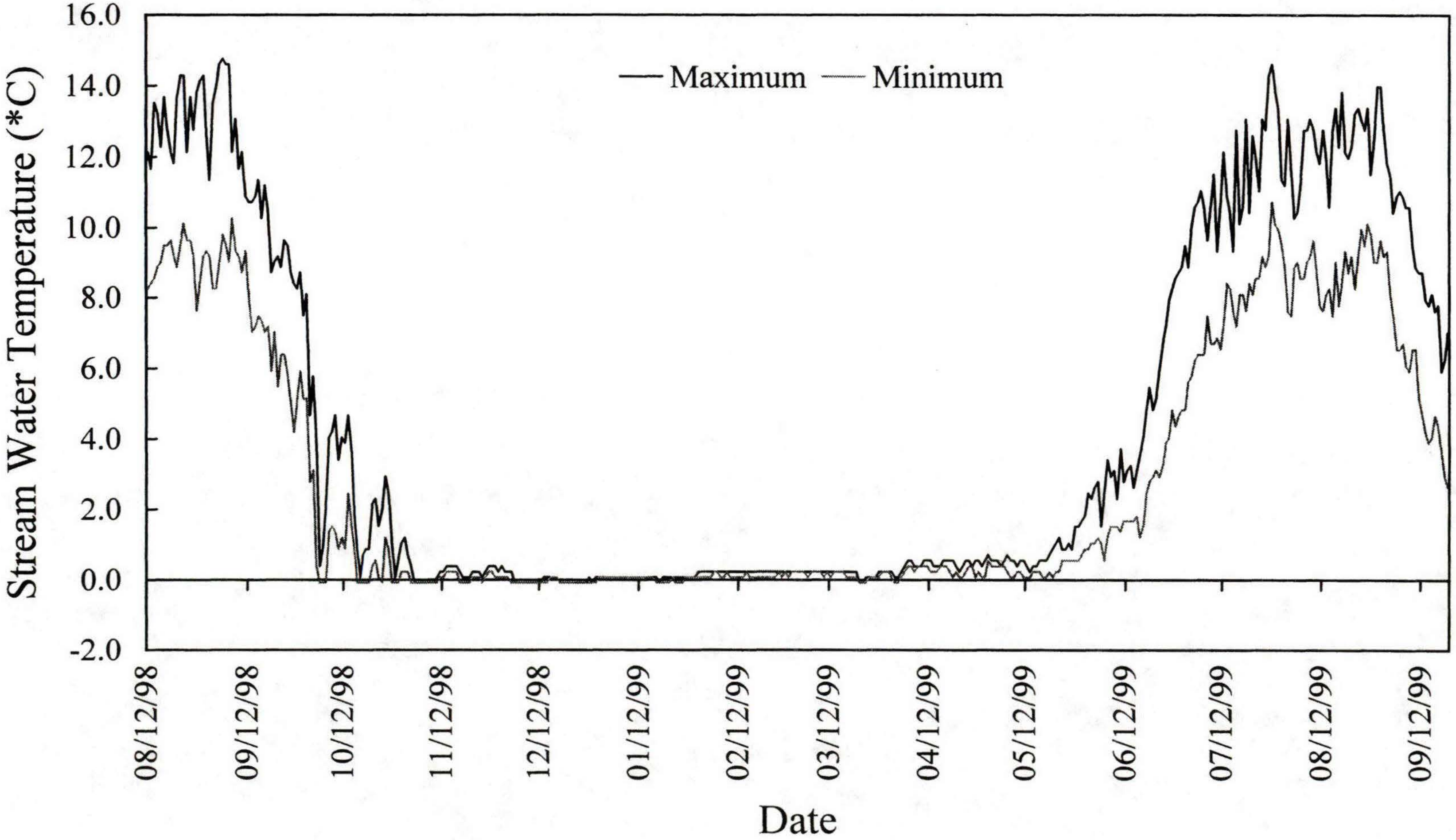



Figure 1. Maximum and minimum daily water temperatures in Cony Creek, 8/12/98 - 9/21/99, measured with an Onset Optic Tidbit thermograph placed in the largest pool.

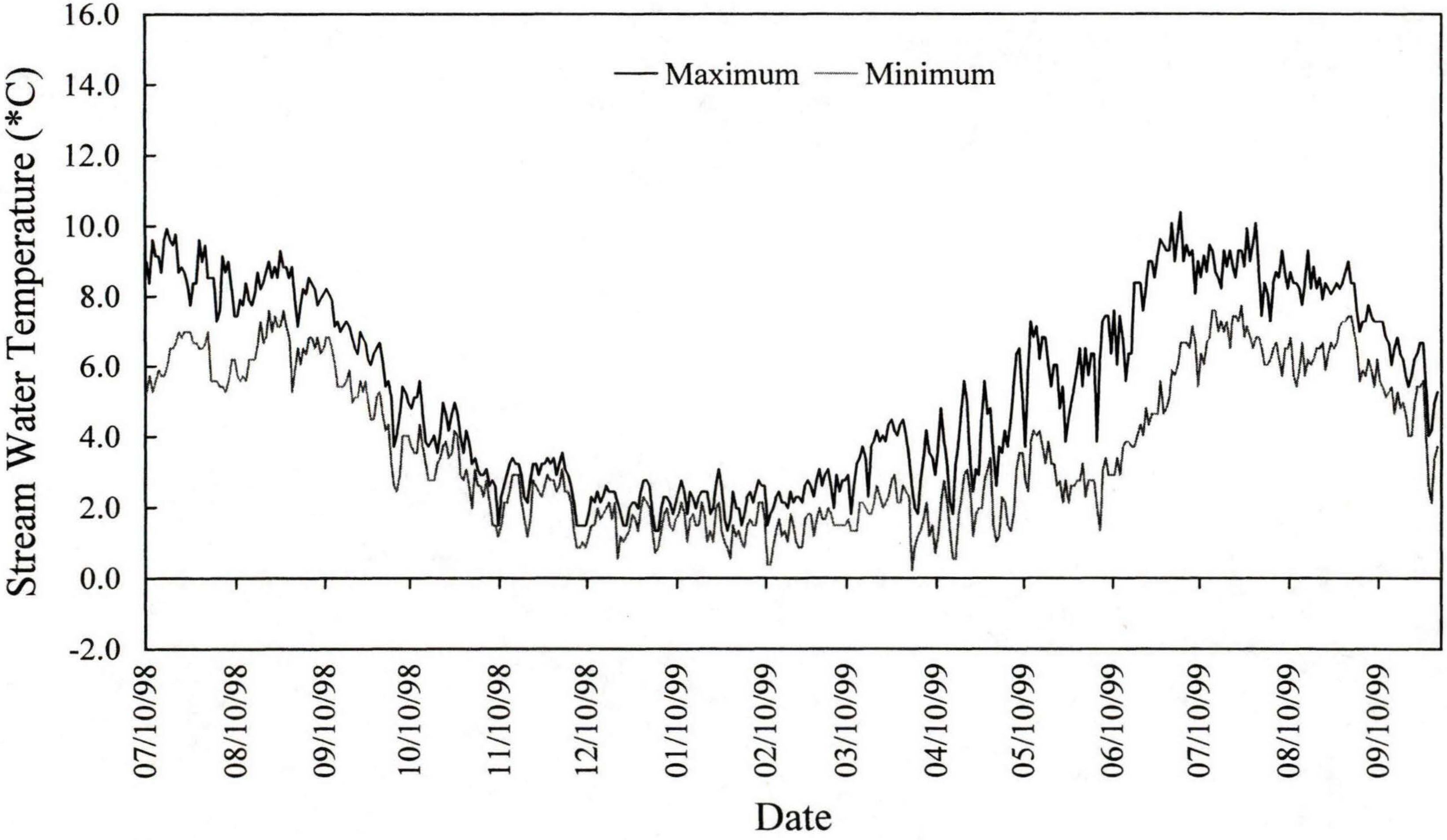



Figure 2. Maximum and minimum daily water temperatures in Cottonwood Creek, 7/10/98 - 10/01/99, measured with an Onset Optic Stowaway thermograph placed in the largest pool.

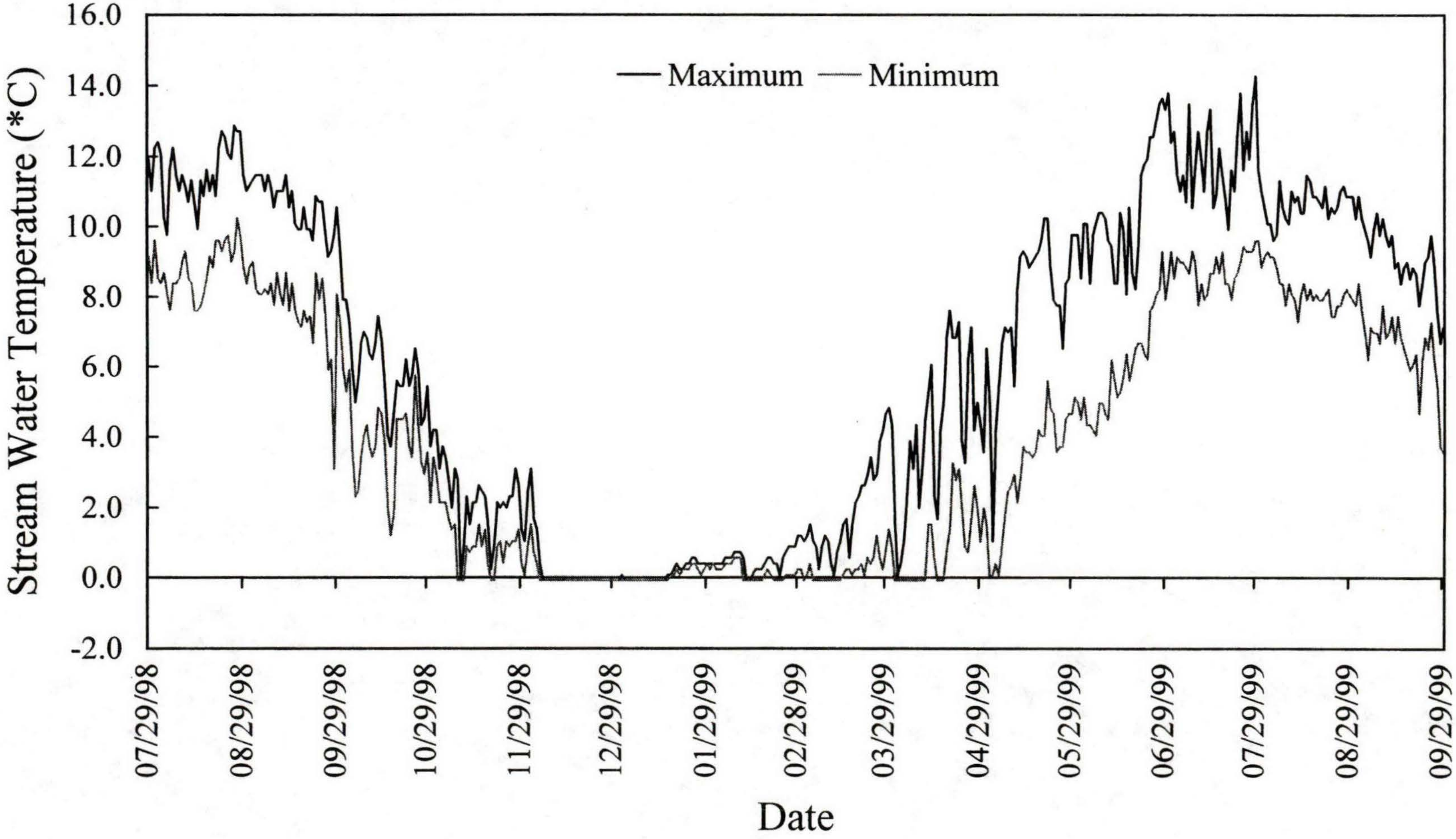



Figure 3. Maximum and minimum daily water temperatures in Doctor Creek, 7/29/98 - 9/30/99, measured with an Onset Optic Stowaway thermograph placed in the largest pool.

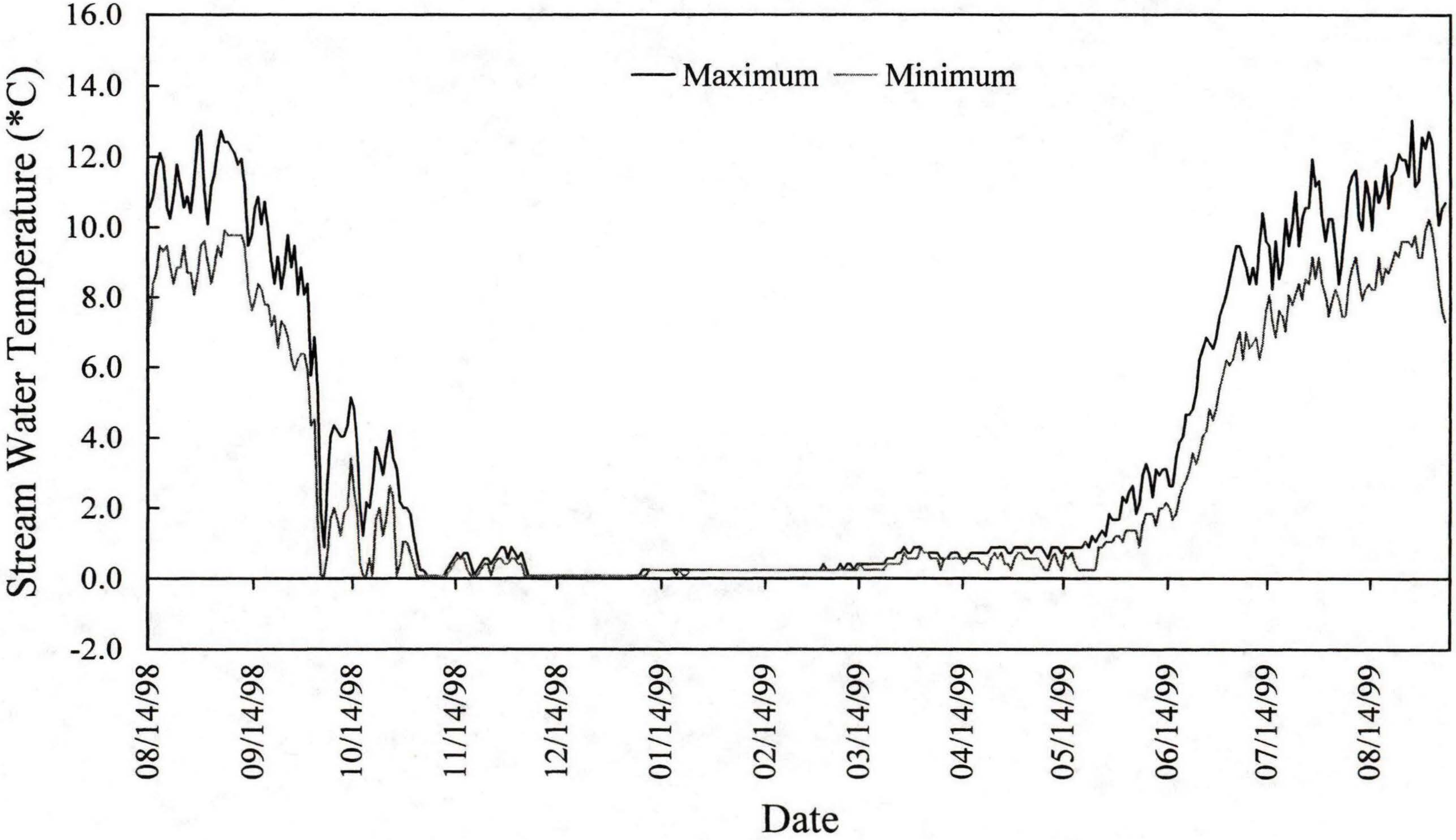



Figure 4. Maximum and minimum daily water temperatures in Fern Creek, 8/14/98 - 9/06/99, measured with an Onset Optic Stowaway thermograph placed in the largest pool.

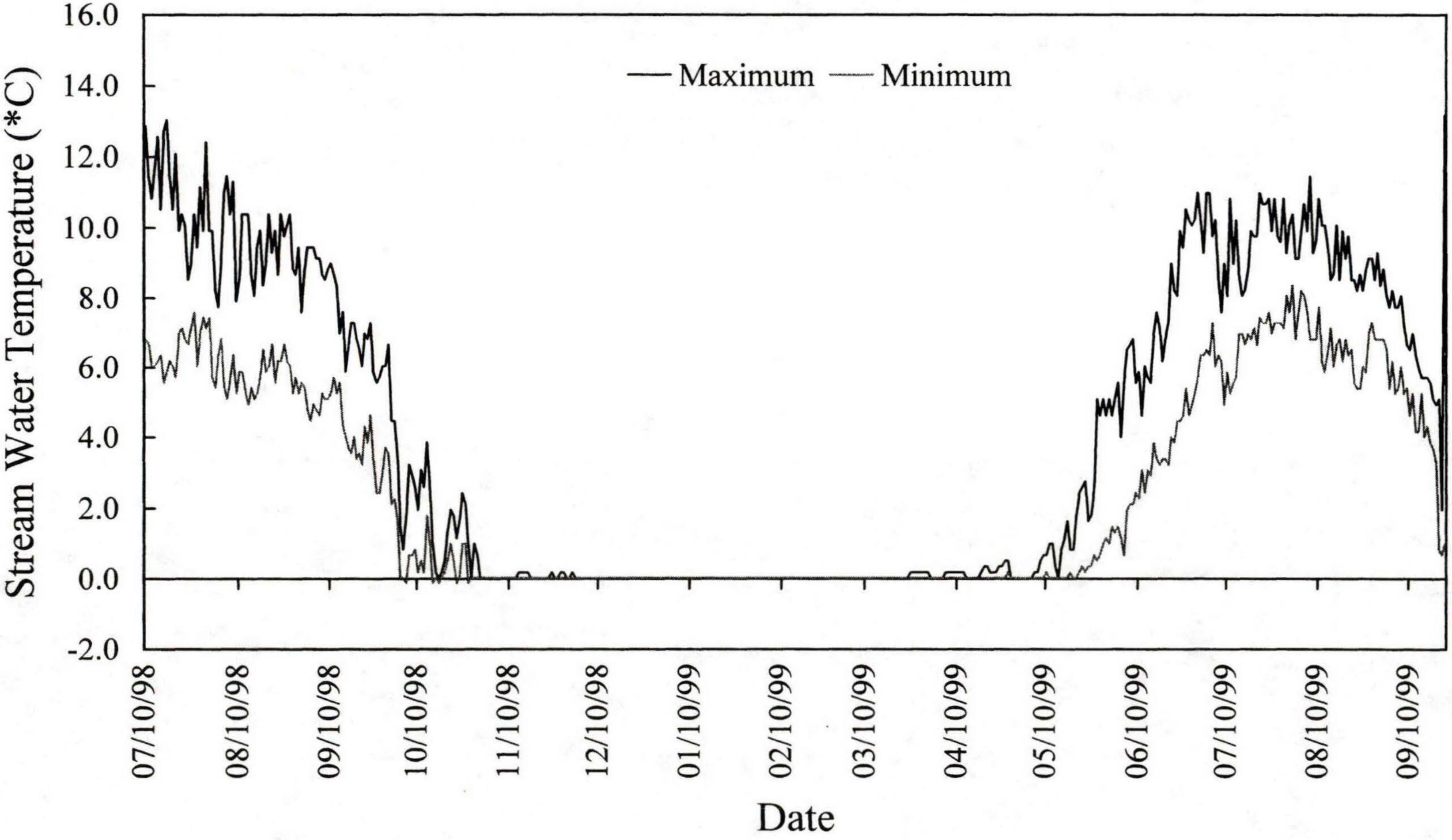



Figure 5. Maximum and minimum daily water temperatures in Greenhorn Creek, 7/10/98 - 9/23/99, measured with an Onset Optic Stowaway thermograph placed in the largest pool.

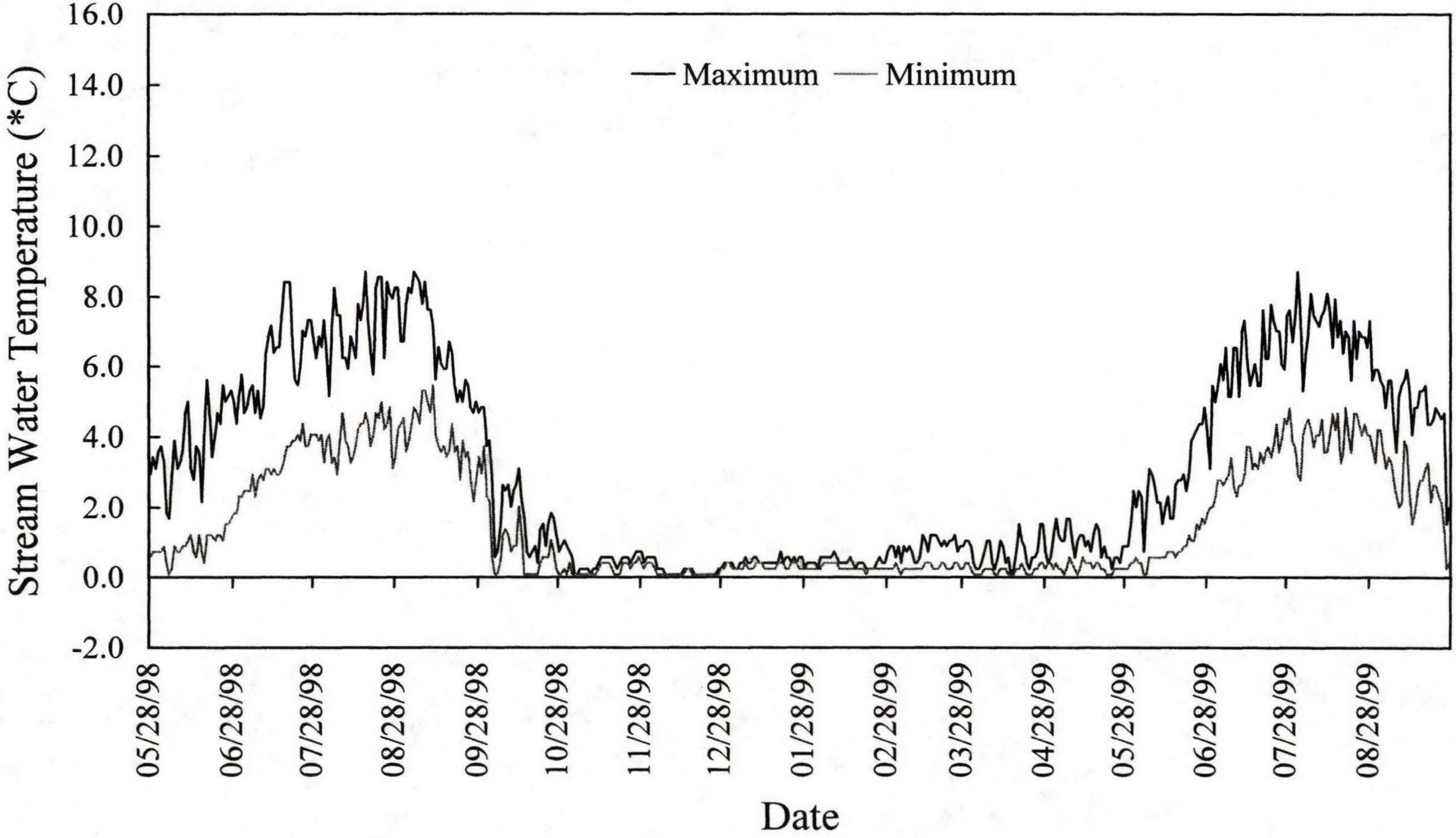



Figure 6. Maximum and minimum daily water temperatures in Hourglass Creek, 5/28/98 - 9/28/99, measured with an Onset Optic Stowaway thermograph placed in the largest pool.

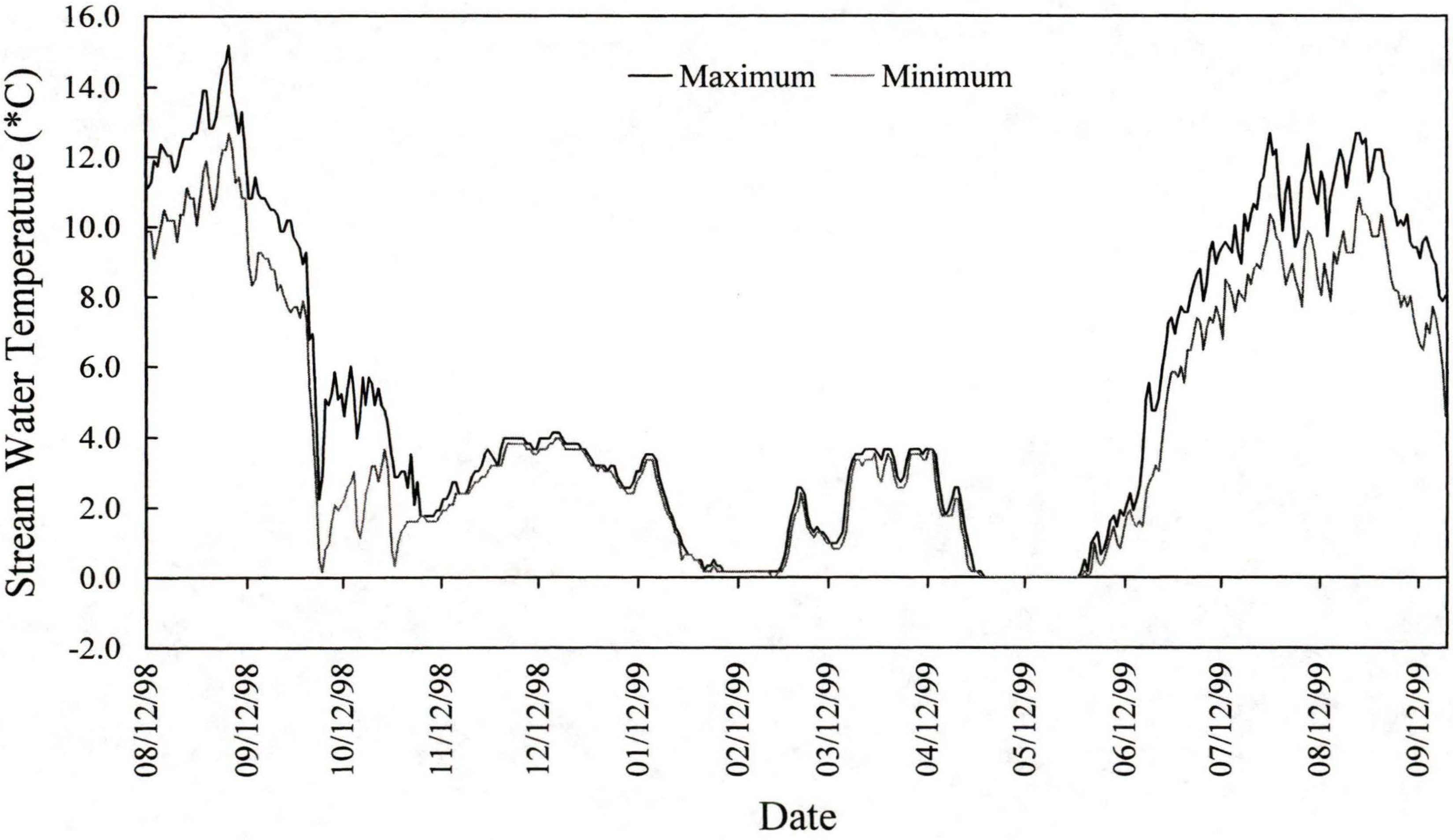



Figure 7. Maximum and minimum daily water temperatures in Lower Hutcheson Lake, 8/12/98 - 9/21/99, measured with an Onset Optic Tidbit thermograph placed along the right bank below the inlet.

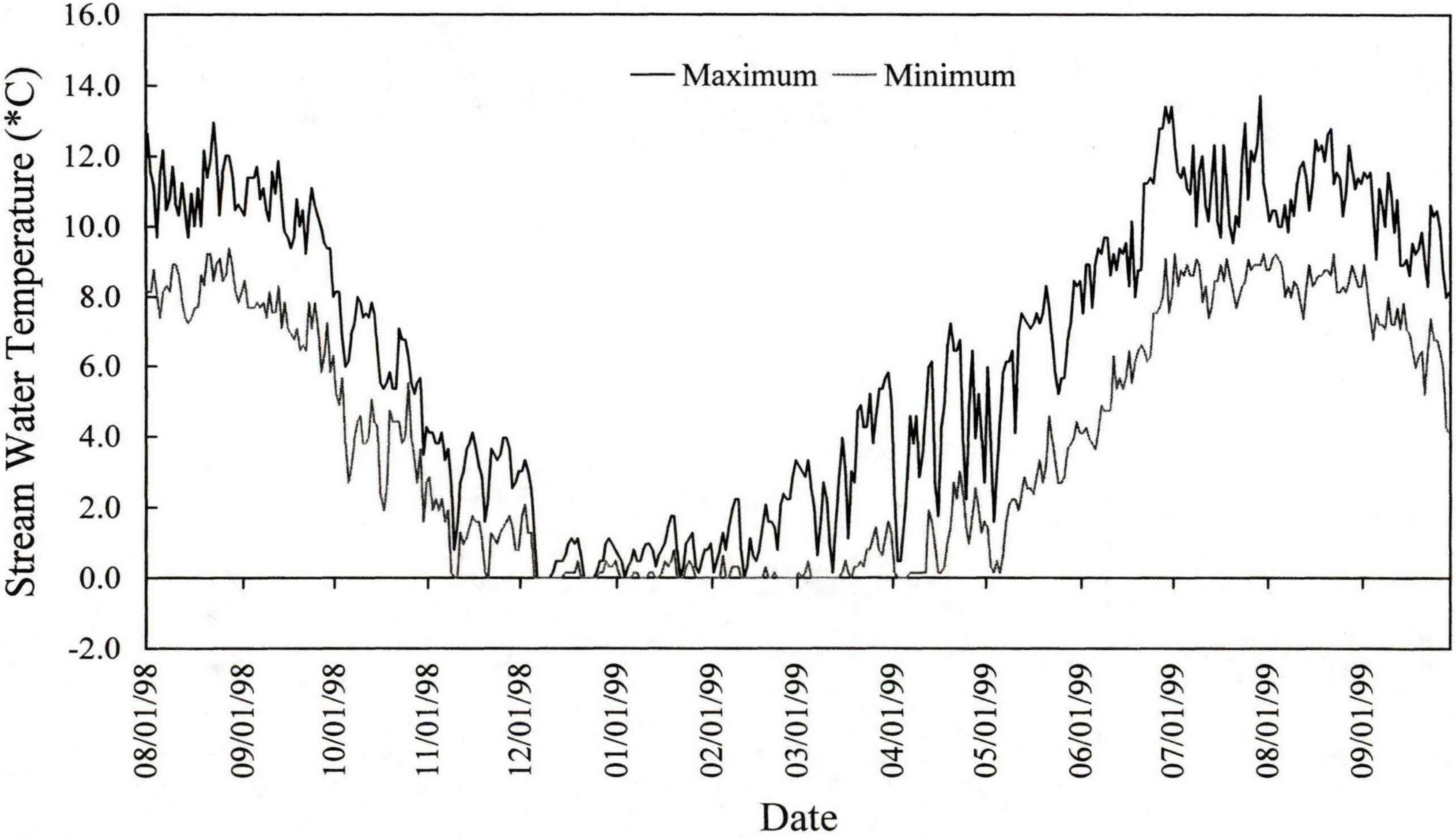



Figure 8. Maximum and minimum daily water temperatures in Jacks Creek, 8/01/98 - 9/30/99, measured with an Onset Optic Stowaway thermograph placed in the largest pool.

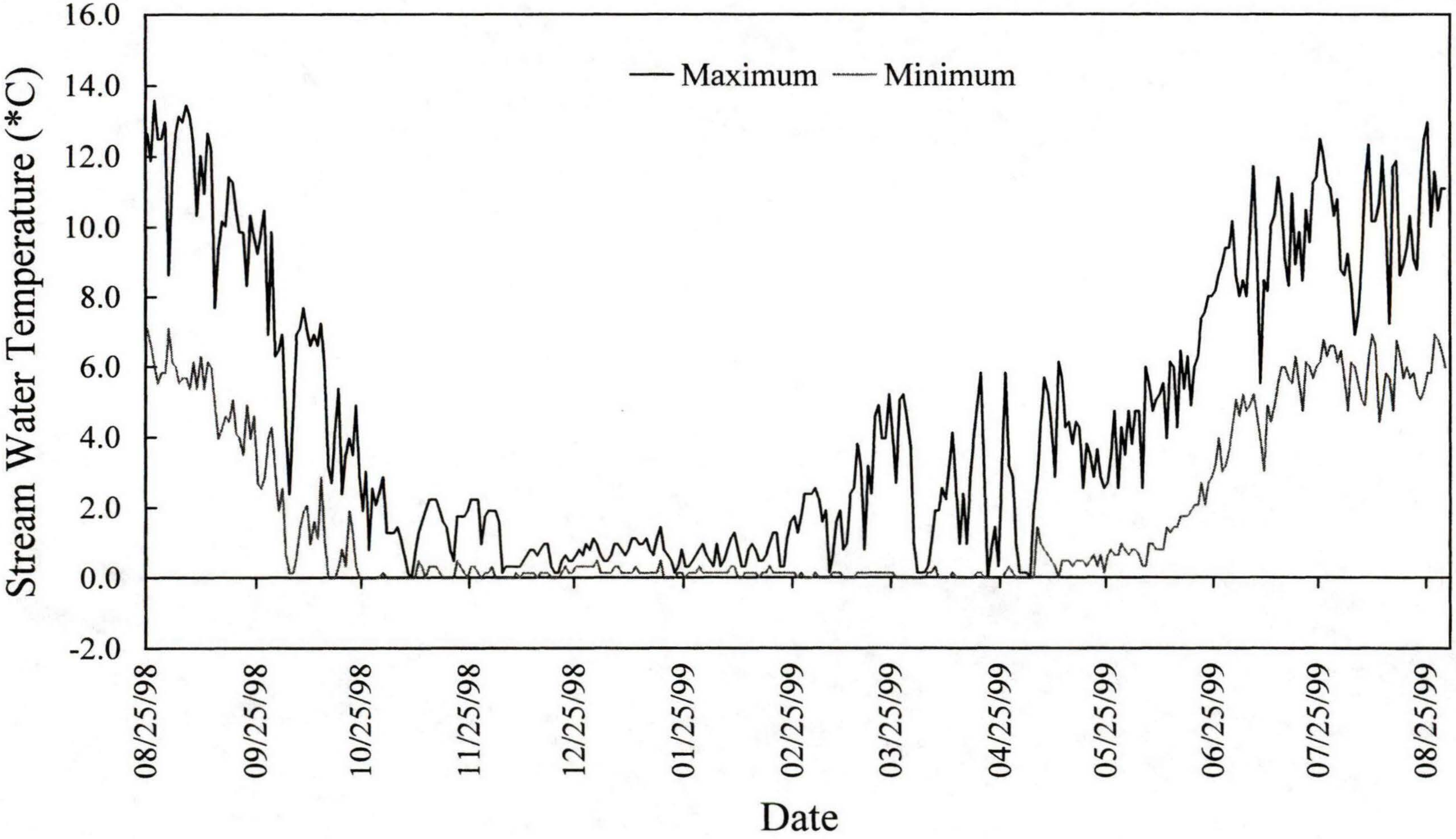



Figure 9. Maximum and minimum daily water temperatures in Little Ute Creek, 8/25/98 - 8/31/99, measured with an Onset Optic Tidbit thermograph placed in the largest pool.

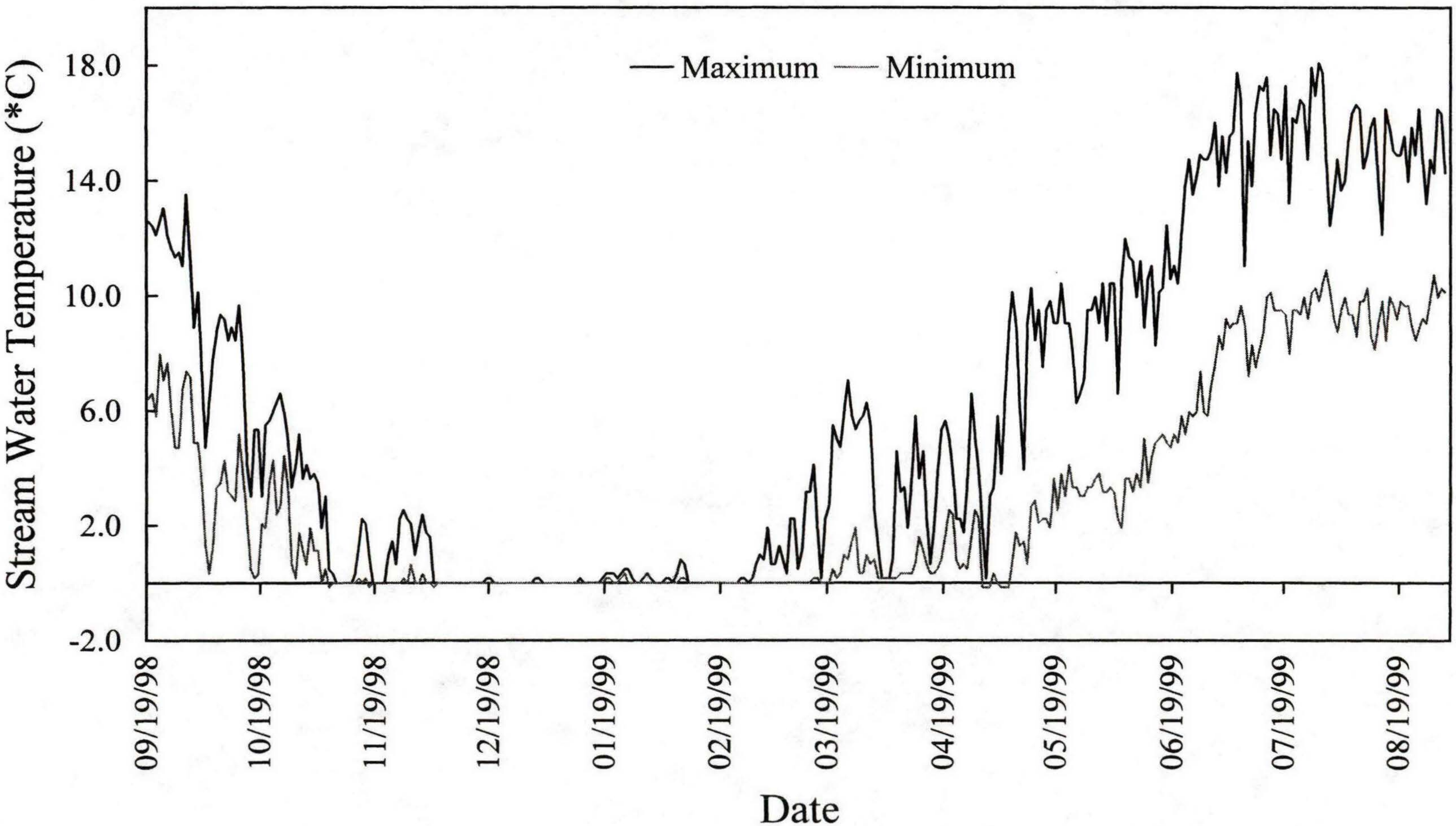



Figure 10. Maximum and minimum daily water temperatures in Medano Creek, 9/19/98 - 9/01/99, measured with an Onset Optic Tidbit thermograph placed in the largest pool.

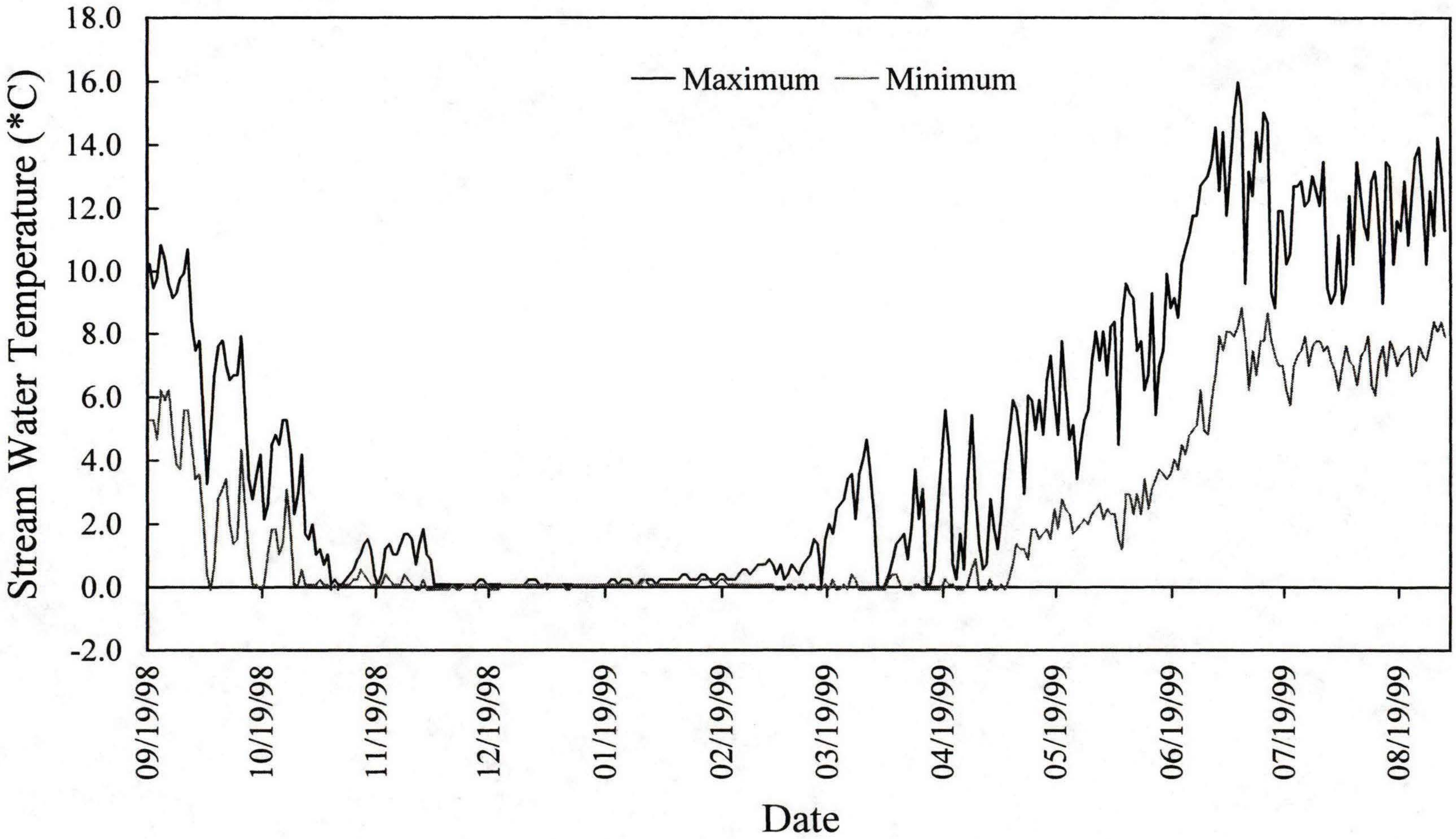



Figure 11. Maximum and minimum daily water temperatures in Medano Creek Hudson Branch, 9/19/98 - 9/01/99, measured with an Onset Optic Tidbit thermograph placed in the largest pool.

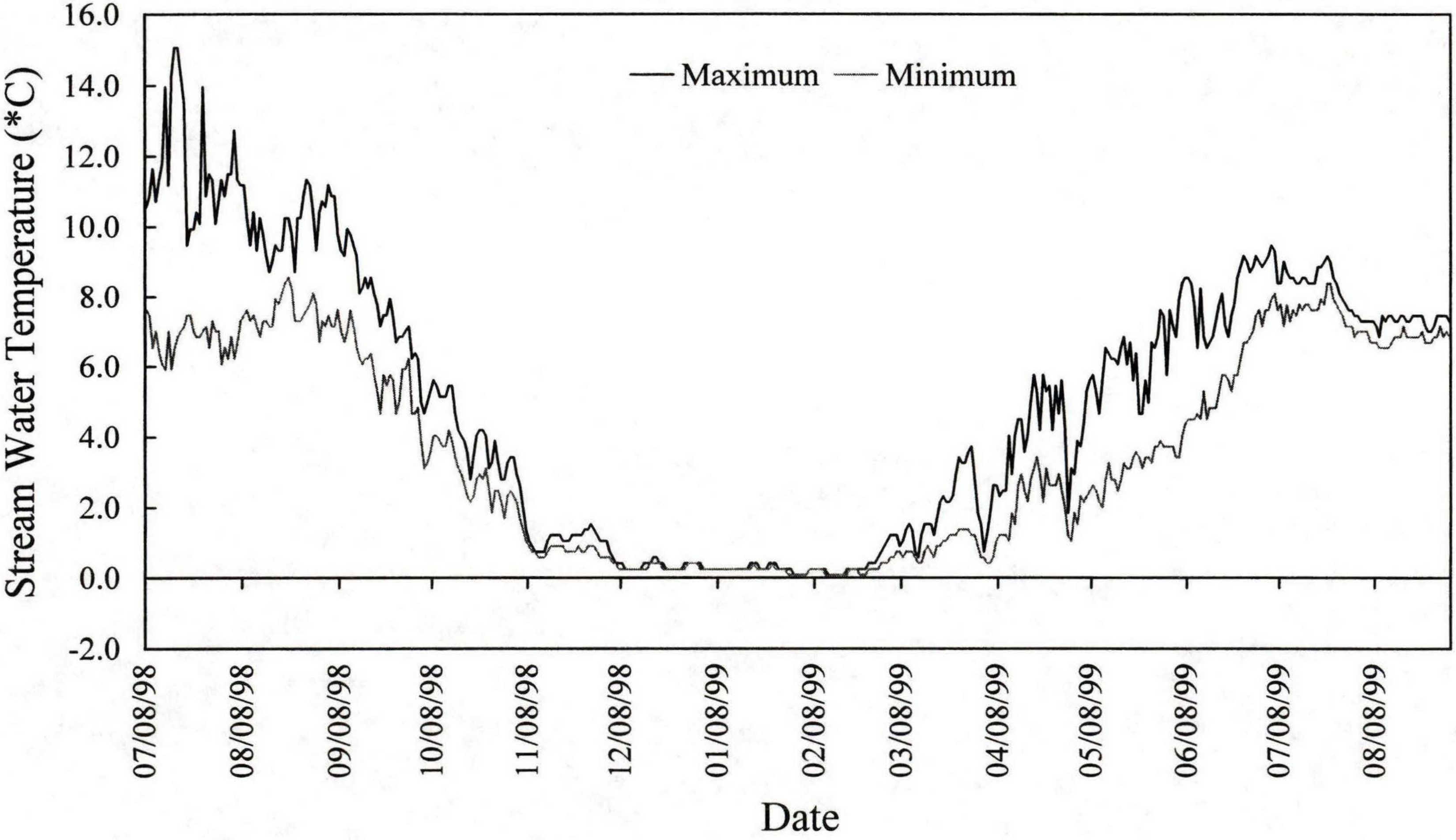



Figure 12. Maximum and minimum daily water temperatures in Middle Fork San Francisco Creek, 7/08/98 - 9/02/99, measured with an Onset Optic Stowaway thermograph placed in the largest pool.

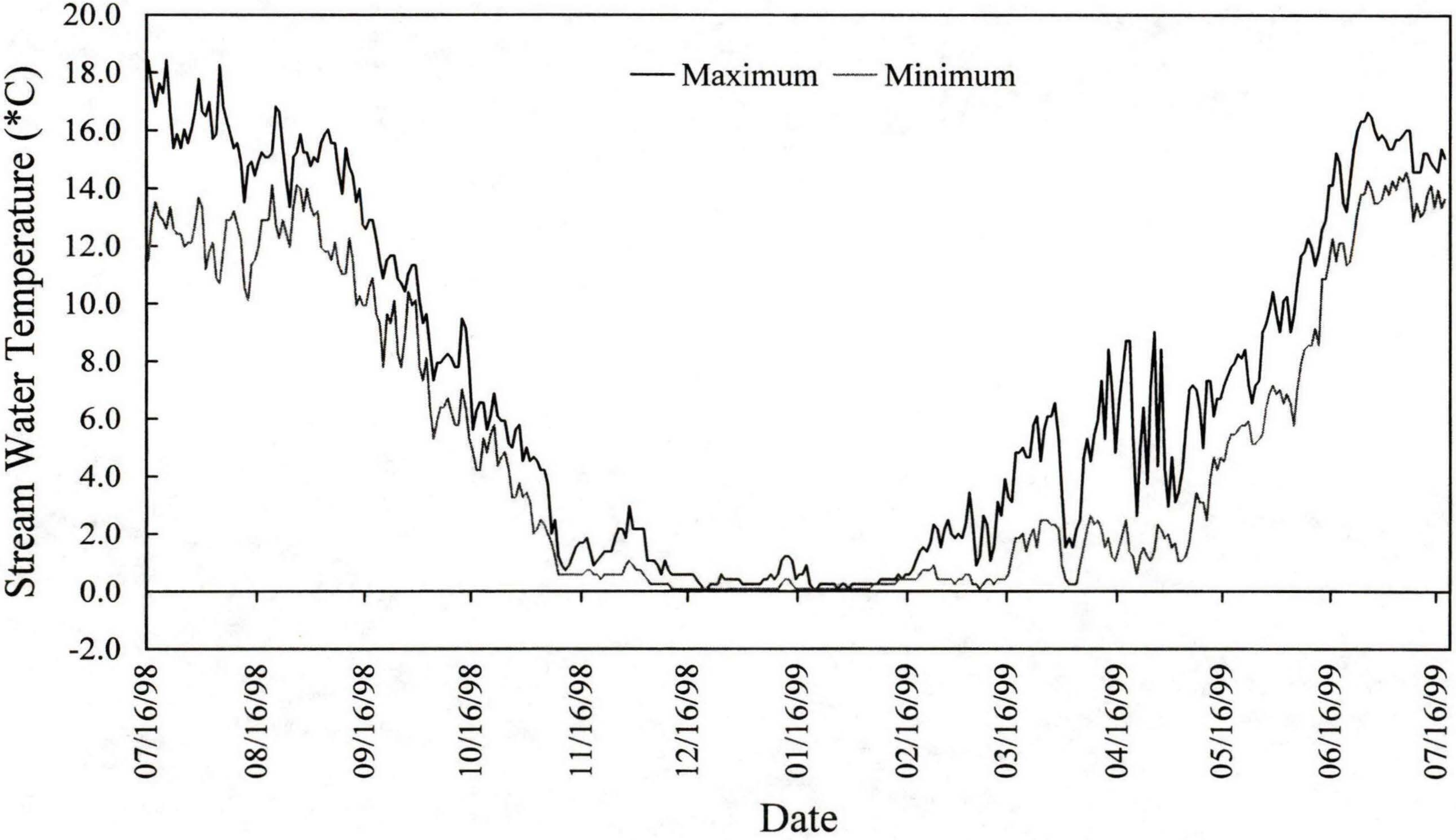



Figure 13. Maximum and minimum daily water temperatures in Nabor Creek, 7/16/98 - 7/19/99, measured with an Onset Optic Stowaway thermograph placed in a beaver pond upstream from Nabor Lake.

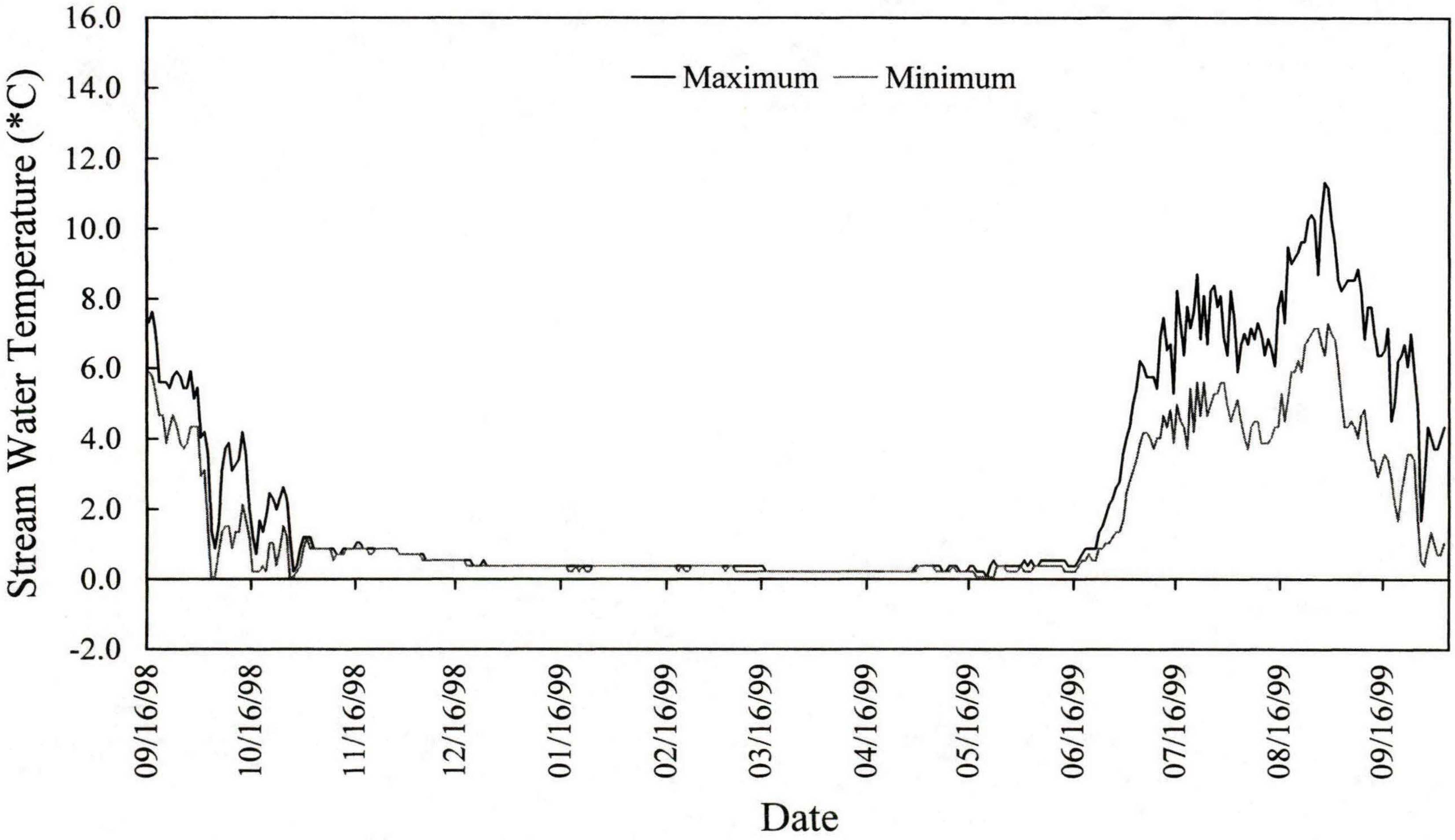



Figure 14. Maximum and minimum daily water temperatures in upper Ouzel Creek, 9/16/98 - 10/05/99, measured with an Onset Optic Tidbit thermograph placed in the largest pool.

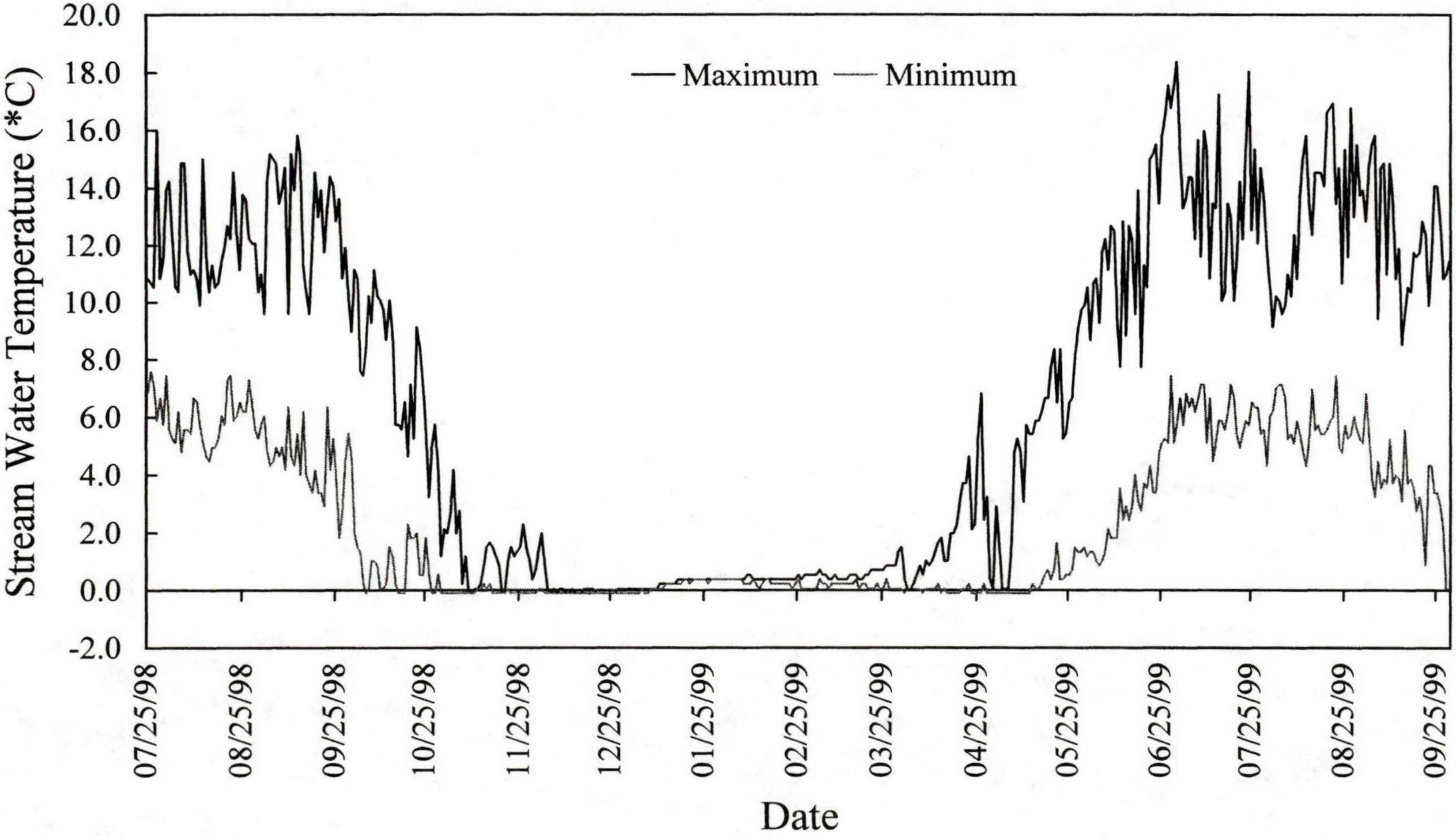



Figure 15. Maximum and minimum daily water temperatures in upper Pecos River, 7/25/98 - 9/30/99, measured with an Onset Optic Stowaway thermograph placed in the largest pool.

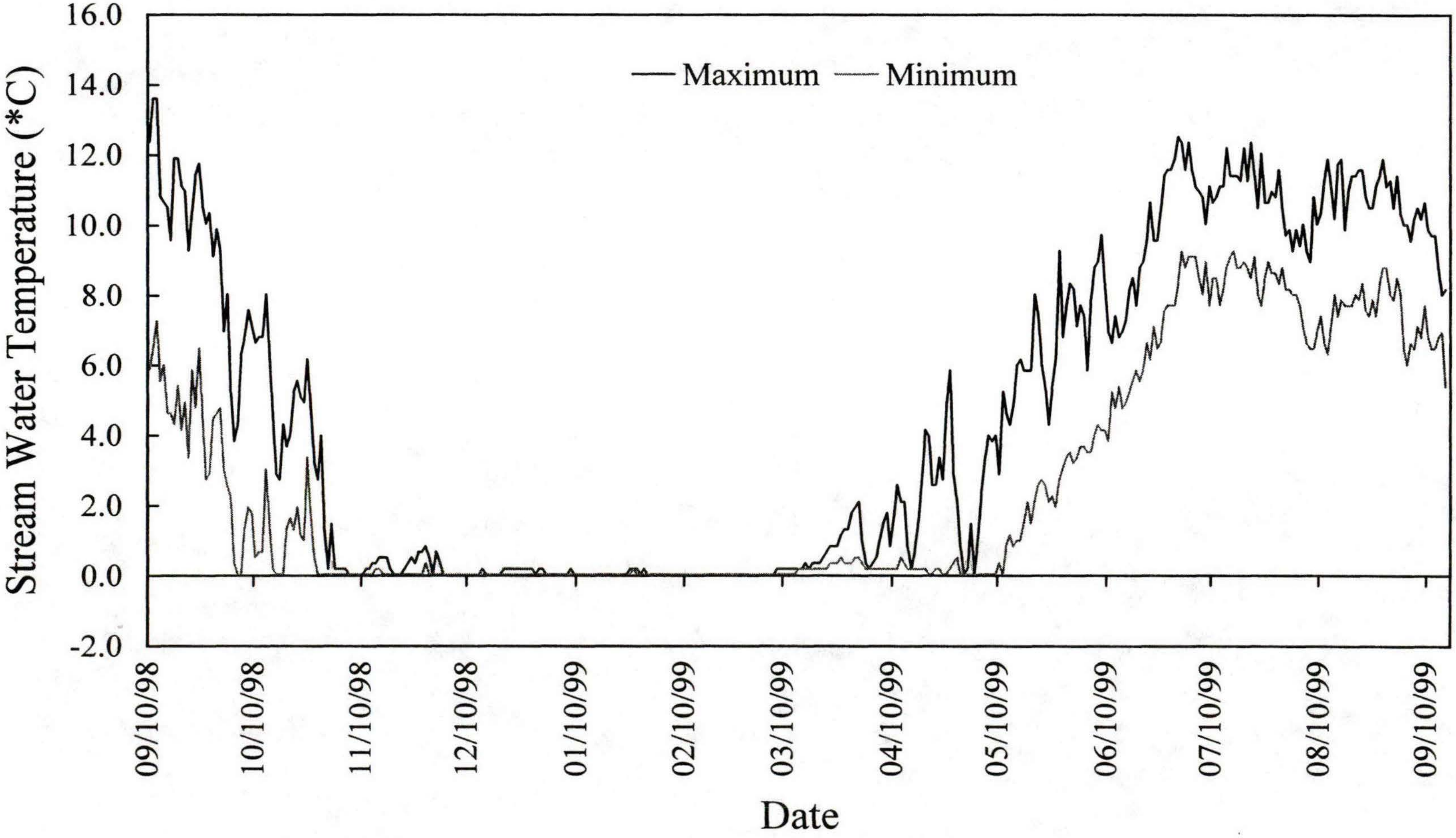



Figure 16. Maximum and minimum daily water temperatures in Powderhouse Creek, 9/10/98 - 9/16/99, measured with an Onset Optic Stowaway thermograph placed in the largest pool.

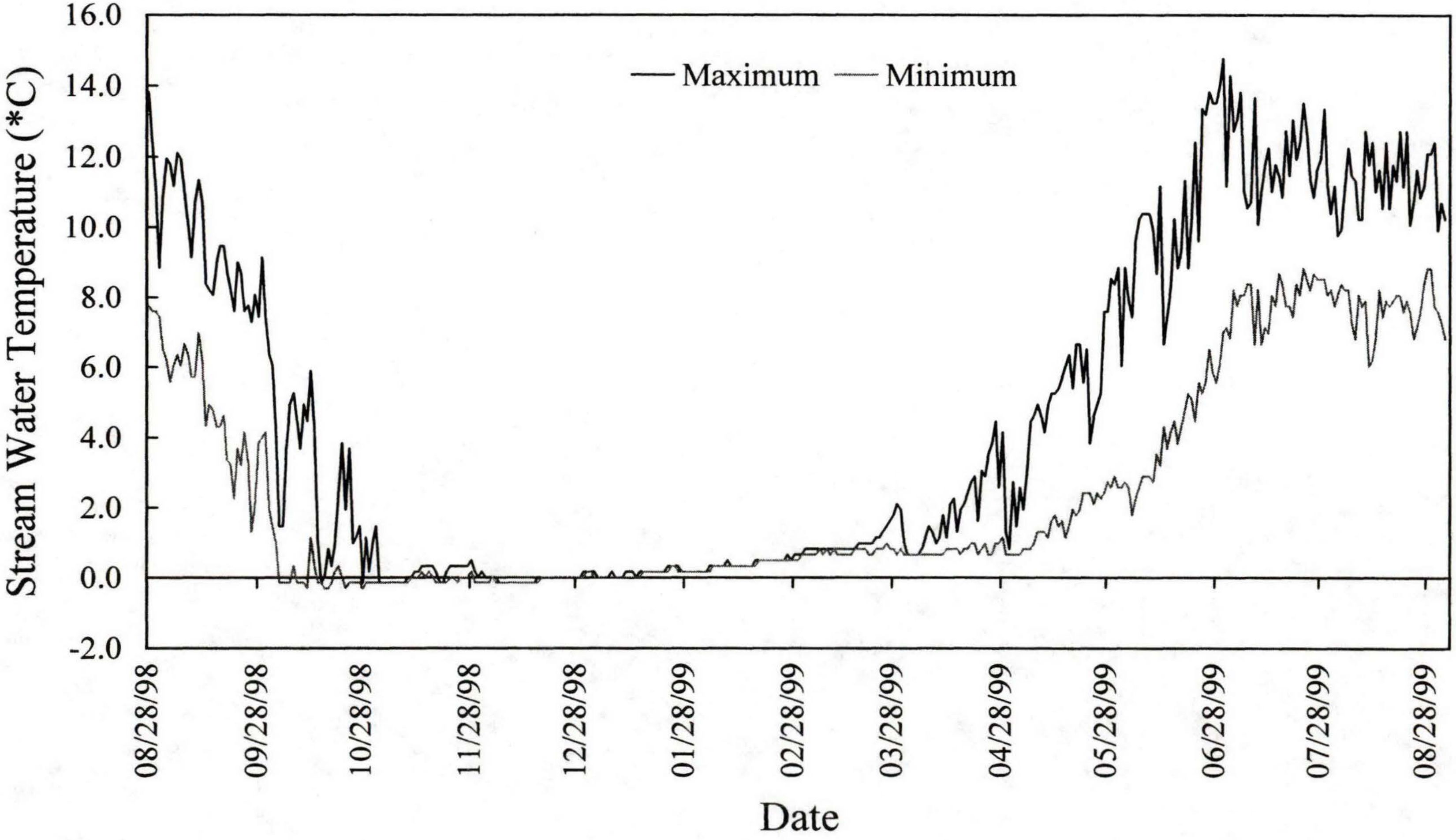



Figure 17. Maximum and minimum daily water temperatures in Rhodes Gulch, 8/28/98 - 9/03/99, measured with an Onset Optic Tidbit thermograph placed in the largest pool.

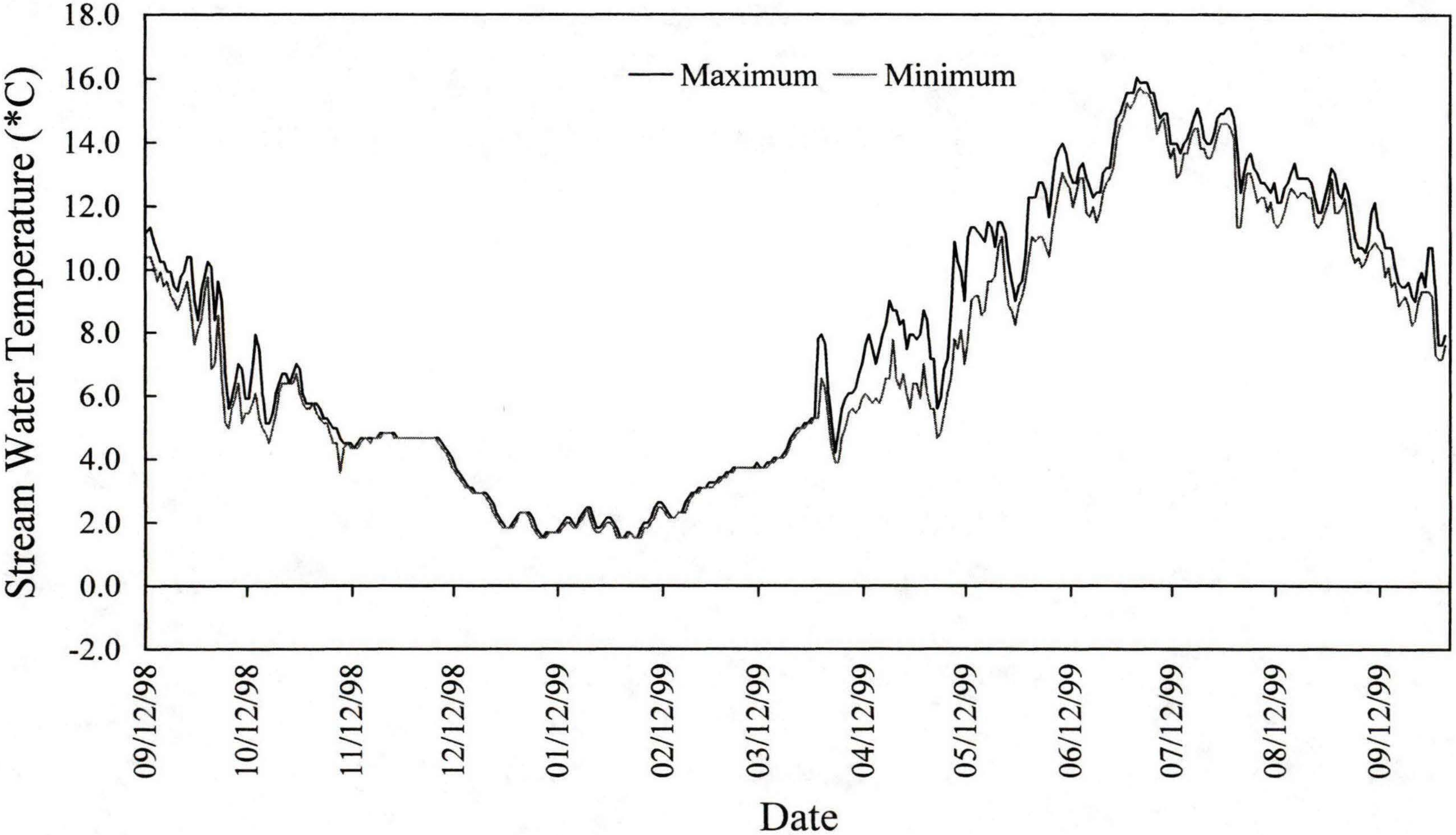



Figure 18. Maximum and minimum daily water temperatures in Rio Cebolla, 9/12/98 - 10/02/99, measured with an Onset Optic Tidbit thermograph placed in the lake just above the dam.

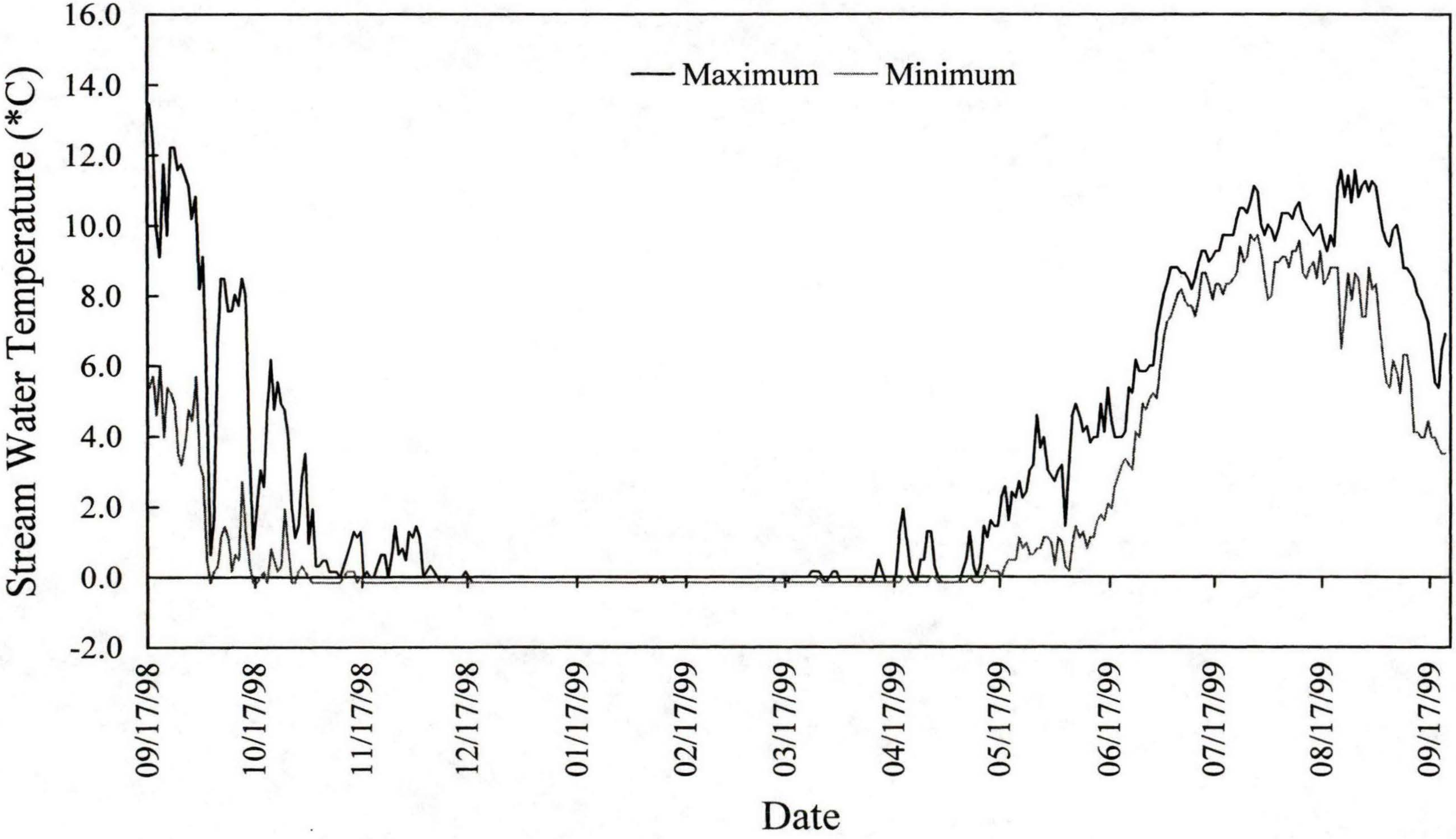



Figure 19. Maximum and minimum daily water temperatures in Roaring River, 9/17/98 - 9/22/99, measured with an Onset Optic Tidbit thermograph placed in the largest pool.

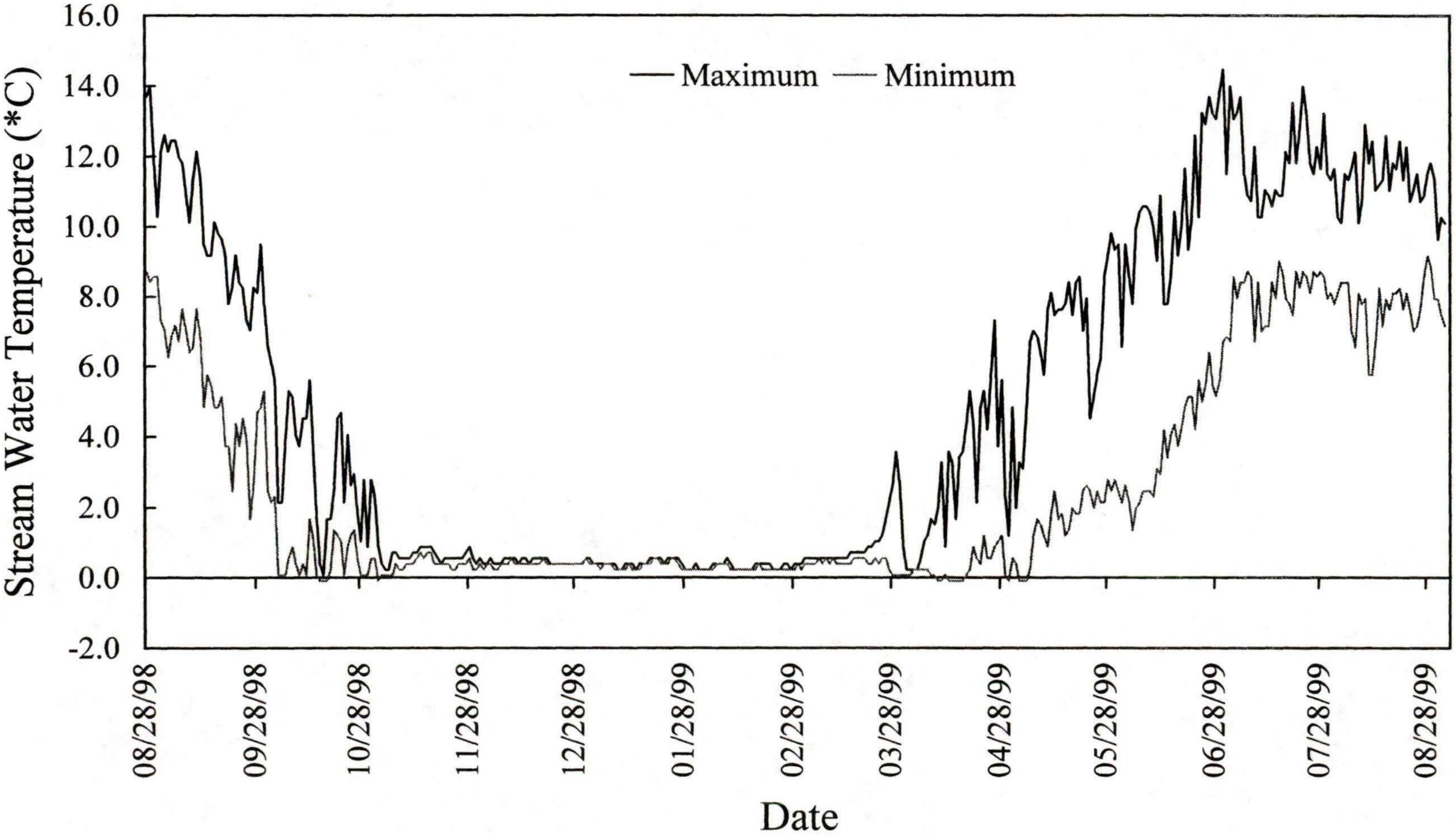



Figure 20. Maximum and minimum daily water temperatures in Rough Canyon, 8/28/98 - 9/03/99, measured with an Onset Optic Tidbit thermograph placed in the largest pool.

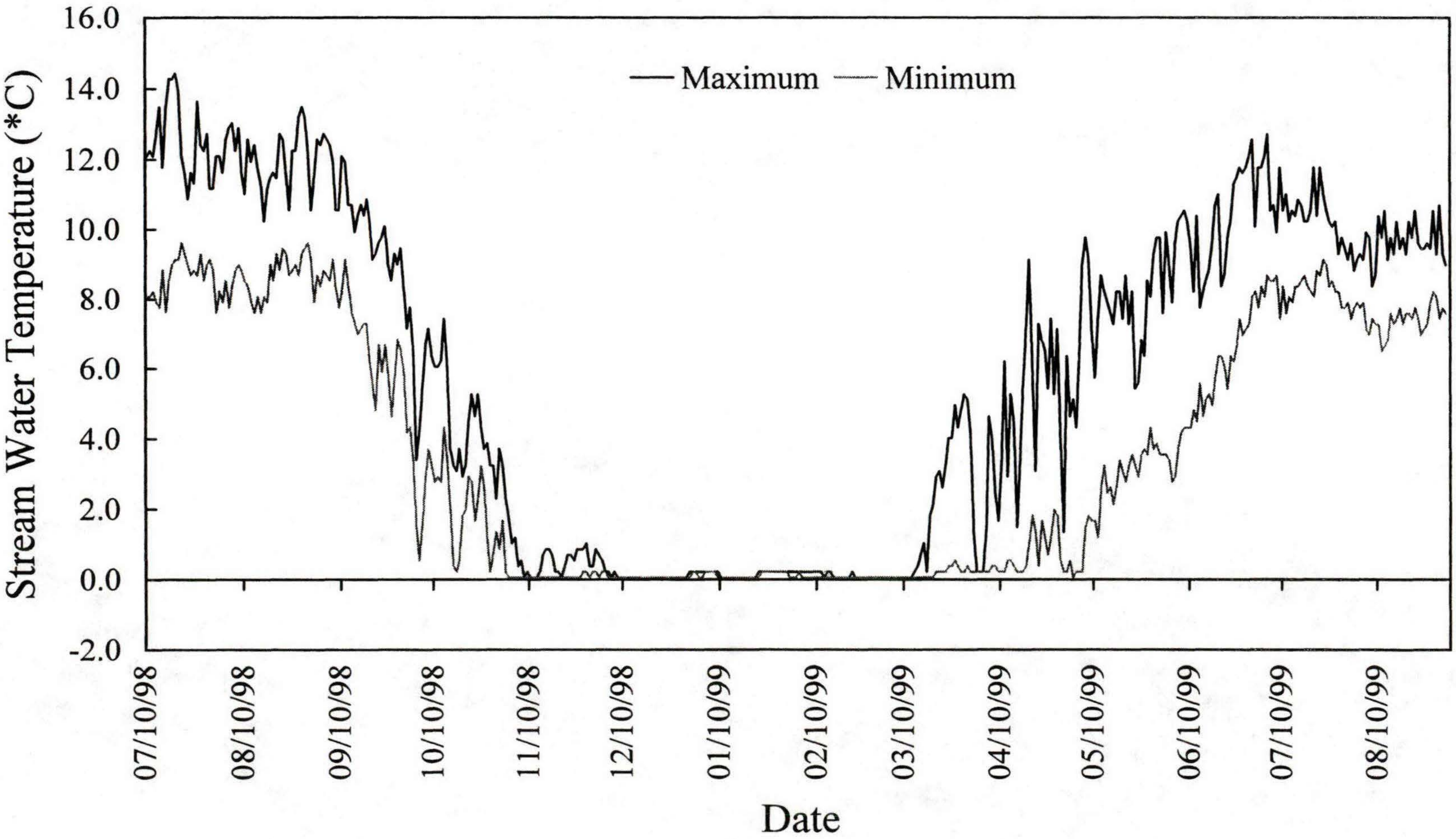



Figure 21. Maximum and minimum daily water temperatures in San Francisco Creek mainstem, 7/10/98 - 9/02/99, measured with an Onset Optic Stowaway thermograph placed in the largest pool.

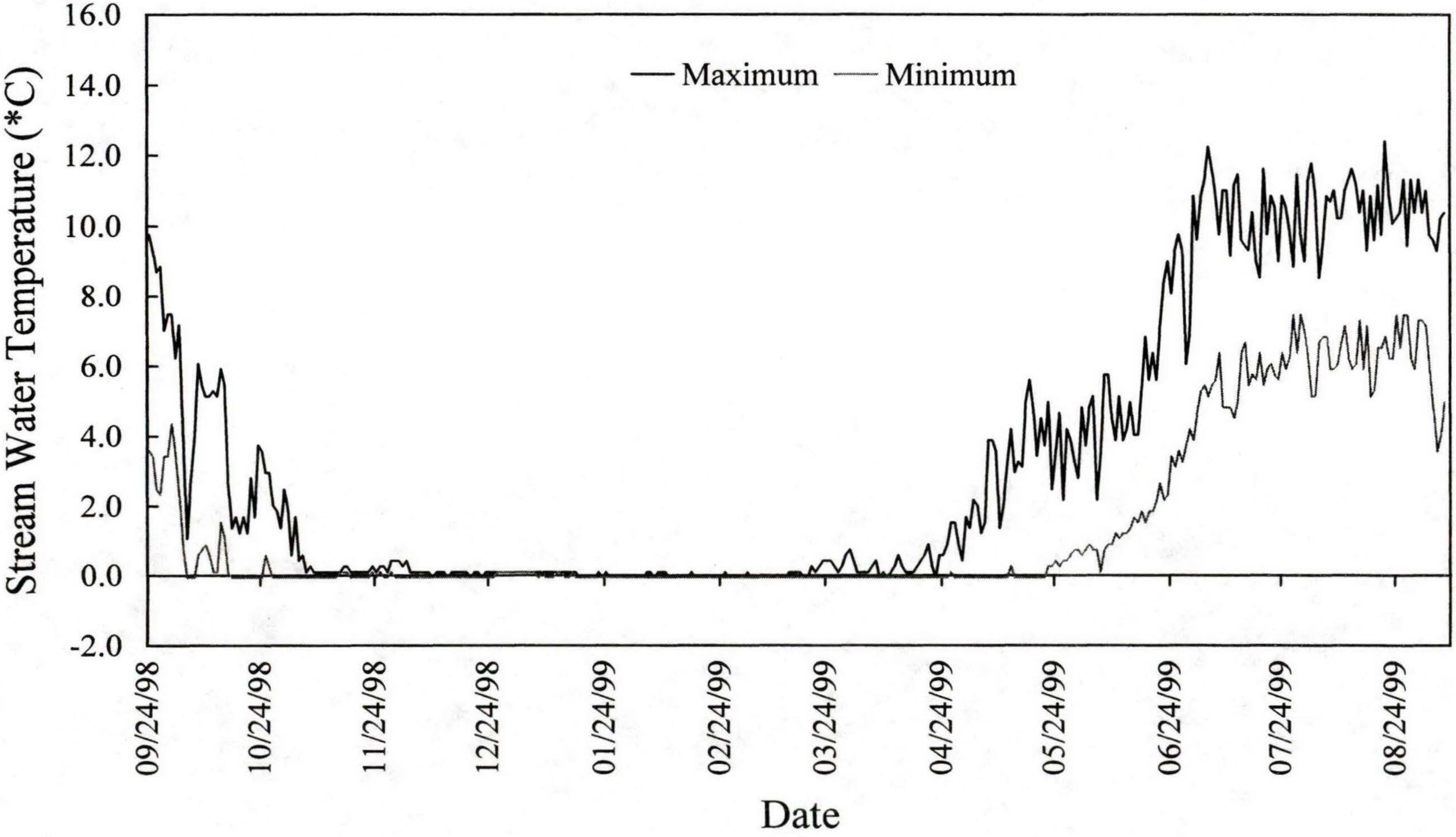



Figure 22. Maximum and minimum daily water temperatures in Sheep Creek, 9/24/98 - 9/07/99, measured with an Onset Optic Tidbit thermograph placed in the largest pool.

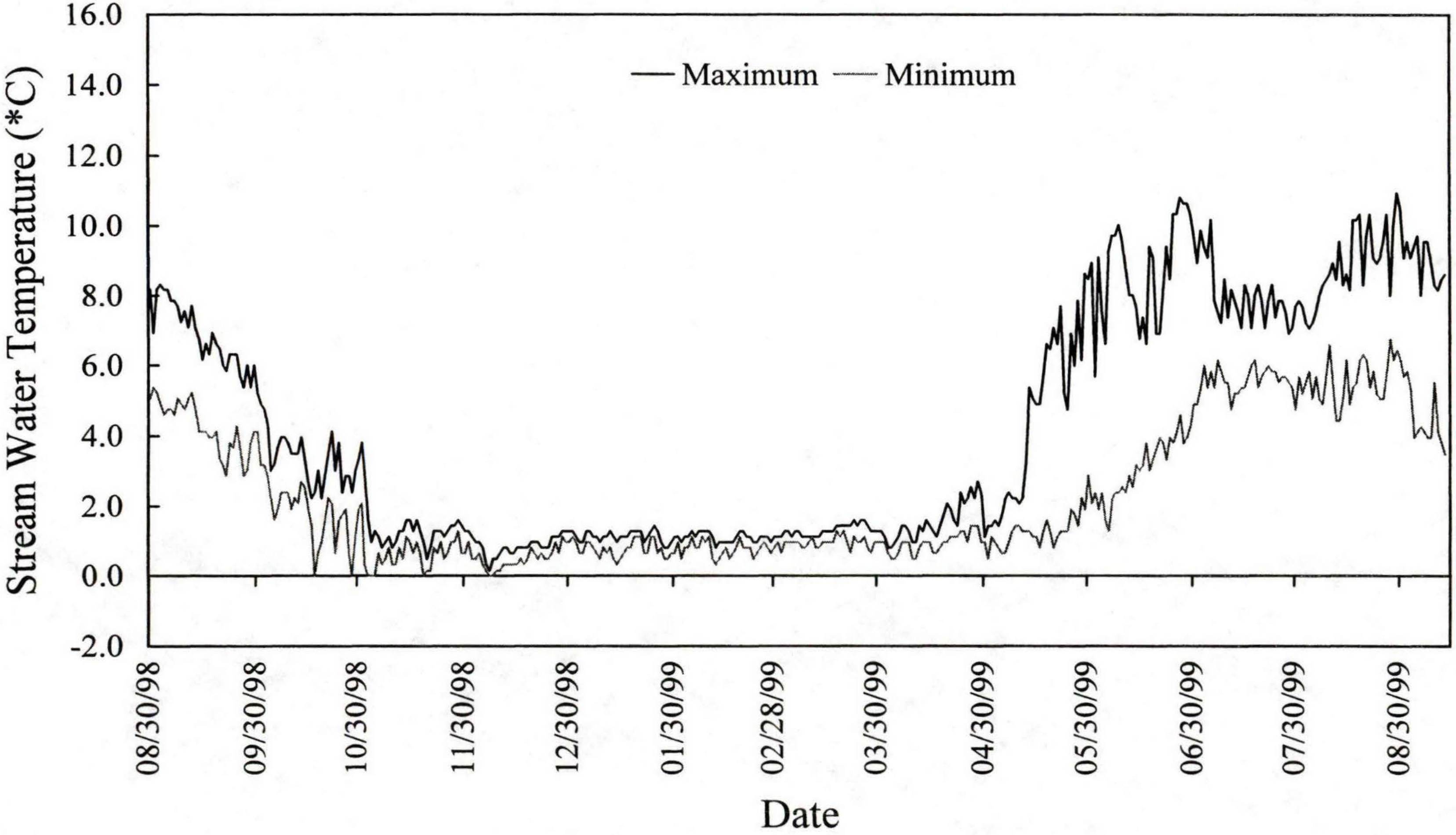



Figure 23. Maximum and minimum daily water temperatures in Unknown Creek, 8/30/98 - 9/13/99, measured with an Onset Optic Tidbit thermograph placed in the largest pool.

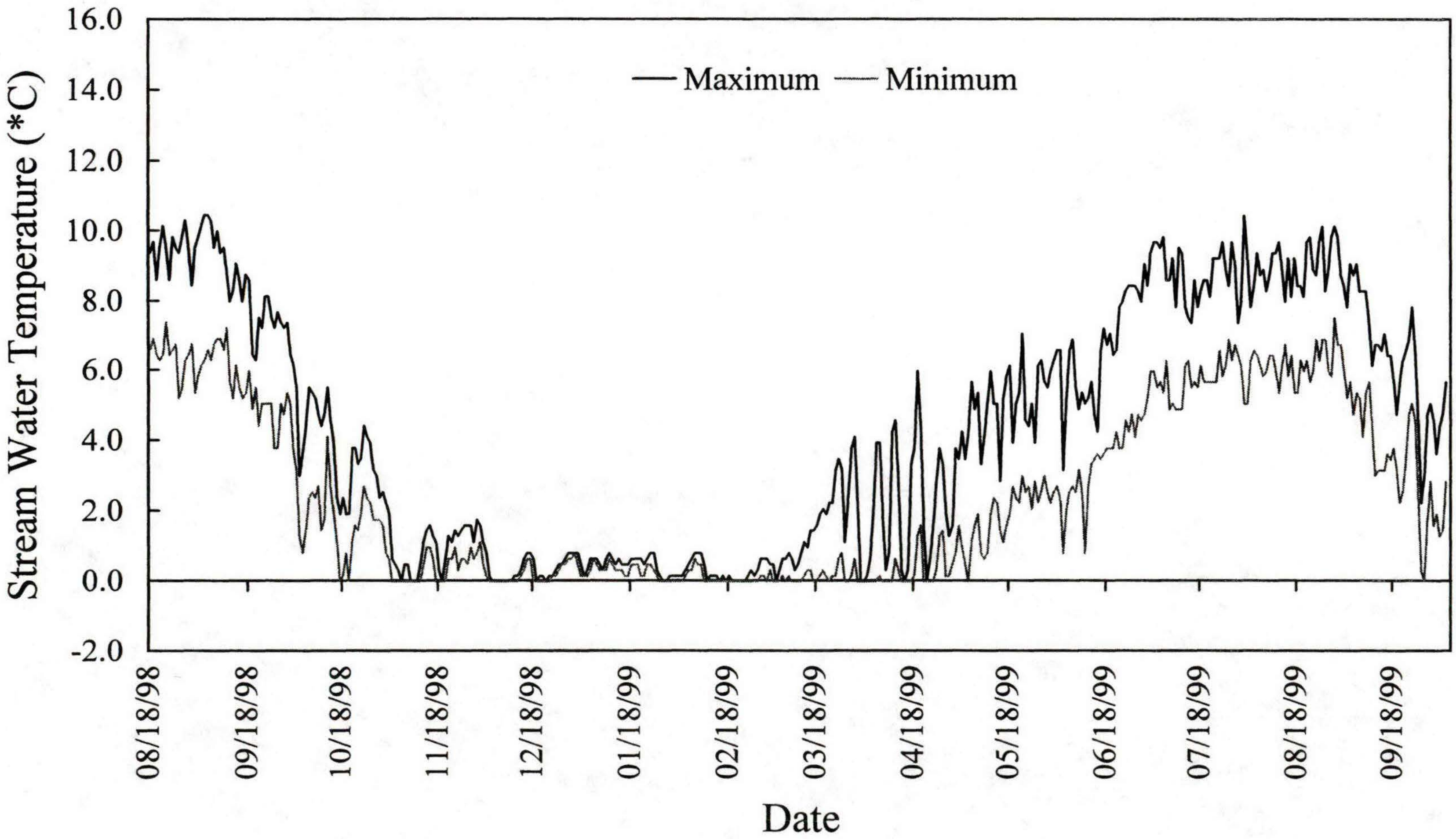



Figure 24. Maximum and minimum daily water temperatures in West Creek, 8/18/98 - 10/06/99, measured with an Onset Optic Tidbit thermograph placed in the largest pool.

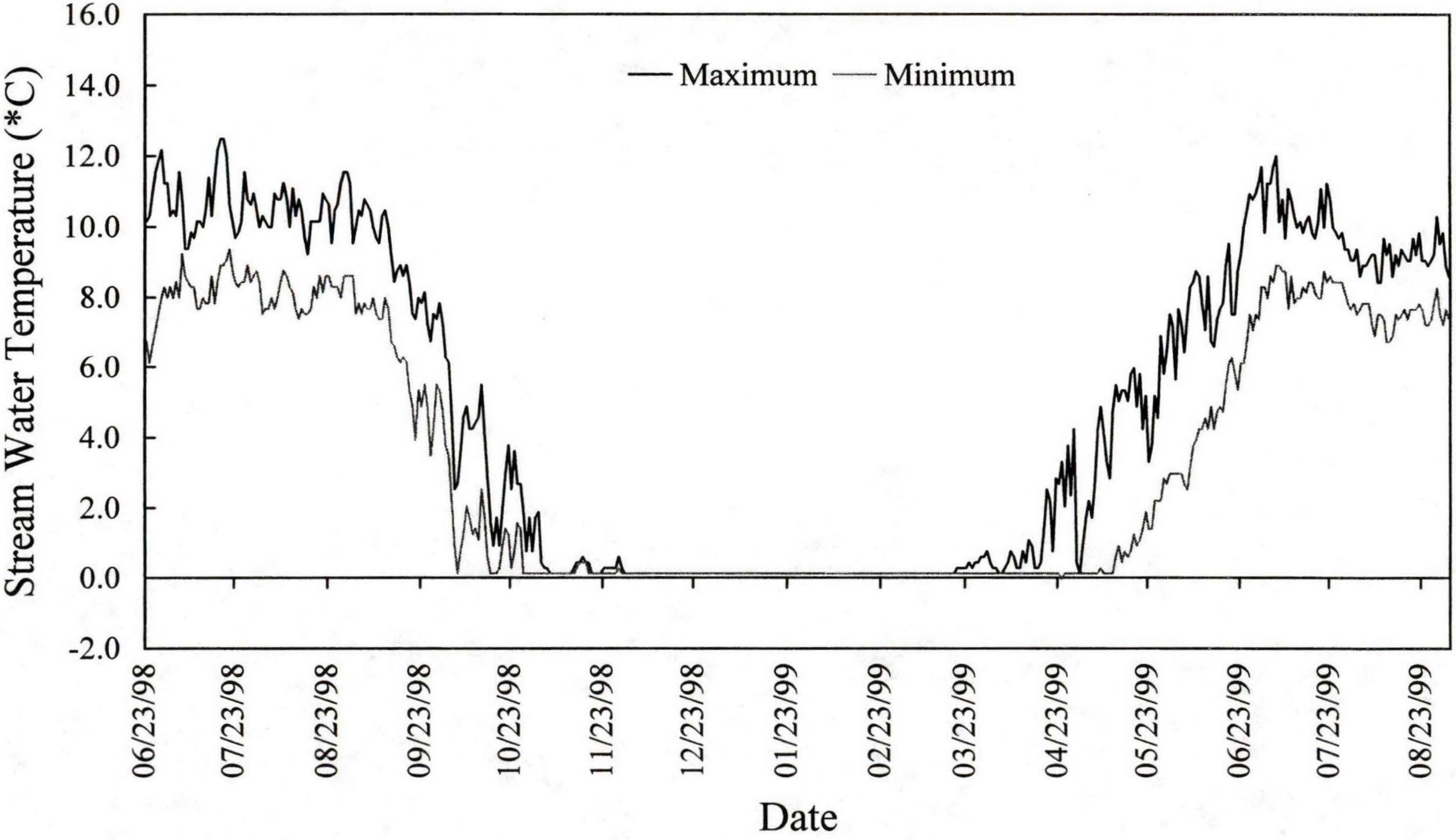



Figure 25. Maximum and minimum daily water temperatures in West Fork San Francisco Creek, 6/23/98 - 9/02/99, measured with an Onset Optic Tidbit thermograph placed in the largest pool.